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ASSESSMENT OF THE POTENTIAL
INCREMENTAL ACID DEPOSITION IN THE
GREATER VANCOUVER AREA FROM THE
BURRARD THERMAL GENERATING PLANT
CSC 110.313-1

Prepared For
BRITISH COLUMBIA HYDRO AND POWER AUTHORITY

Prepared

by

Concord Scientific Corporation

Robert J. Vet
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April, 1983

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SUMMARY

This report presents the results of an assessment of the potential environmental impact of the operation of B.C. Hydro's Burrard Thermal Generating Plant (BTGP) with regard to acid compound deposition in the area. BTGP is a natural gas fired generating station with a rated capacity of 912.5 MW, situated in Port Moody, B.C., at the eastern end of the Burrard Inlet. Natural gas is well known to be a relatively clean burning fuel. The emittants which can potentially contribute to acidic deposition are the oxides of nitrogen. Essentially no sulphur dioxide is emitted.

The assessment was based on an examination of the emission rate of nitrogen oxides from BTGP relative to the total emissions of various pollutants in the GVRD, incorporating relevant meteorological and climatological data. A simple model of chemical transformation and pollutant deposition was applied. Care was taken to use conservative assumptions throughout the assessment process. It is therefore believed that the estimates obtained for the long term incremental effect of BTGP on acid deposition in the Vancouver area represent an upper limit. Even so, it was found that this incremental effect is small; from 0 to 0.4 % for dry deposition and from less than 1 % to 5 % for wet deposition. These percentage increases represent small increments on existing deposition rates which are themselves much less than other areas of Canada (e.g. Southern Ontario).

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1. INTRODUCTION

British Columbia Hydro and Power Authority maintains a thermal generating plant near Port Moody in the Burrard Inlet. They are planning to obtain a permit to operate this plant, known as the Burrard Thermal Generating Plant (BTGP), using natural gas. They have therefore commissioned an assessment of potential environmental effects, one aspect of which is the potential for incremental increases in acid compound deposition in the area. Other concerns include such items as health effects, corrosion, phytotoxicity and ambient air quality. This report primarily deals with the question of the incremental effects of acid compound deposition which may be expected in the Greater Vancouver area attributable to operation of the BTGP. It should be noted that the only emittants from BTGP which can potentially contribute to acidic deposition are the oxides of nitrogen. Combustion of natural gas produces essentially no sulphur dioxide.

1.1 Statement of Work Programme

The objective of this work programme was to review existing information relevant to acid deposition in the Greater Vancouver Regional District airshed and to assess the potential incremental contribution of BTGP emissions.

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1.2 Approach to Assessment

Given the foregoing objective and terms of reference, it was determined that the most appropriate approach to the assessment was as follows:

1. Identify and describe the important physical and chemical processes relating to acid depositon from local sources;

2. Analyse pertinent data applicable to those processes specific to the BTGP and its local environment;

3. Assess the impact of BTGP emissions using (1) a knowledge of the processes involved, (2) a knowledge of previous research and experience in the field and (3) the predictions of a semi-quantitative model to estimate the magnitude of the incremental effect due to BTGP emissions. In this model, a number of conservative simplifying assumptions were used such that the predictions represent upper limits of BTGP's incremental impact.

Chapter 2 of this report identifies and discusses the important atmospheric processes in detail. This provides a basis from which various assumptions and conservative simplifications are applied to the assessment in Chapter 4. Chapter 3 discusses available data from

the Greater Vancouver area with an emphasis on how these data can be usefully applied to the important processes described in Chapter 2. Finally, Chapter 4 presents the assessment in two parts. The first part is a qualitative assessment based on a knowledge of the emissions, the important atmospheric processes, and previous field research. The second part is a semi-quantitative assessment based on the results of the aforementioned model.

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2. OVERVIEW OF IMPORTANT ATMOSPHERIC PROCESSES

In a combustion process, such as that occurring in a power plant, various gaseous pollutants are produced; among these, sulphur dioxide and the oxides of nitrogen are predominant. These species are emitted from the stack, and are transported by air movements, during which process there exists the potential for chemical reaction, and the resulting production of secondary pollutants. Mechanisms exist which are capable of depositing both primary and secondary pollutants on the earth's surface and these deposition processes may be subdivided into wet deposition, which occurs during precipitation, and dry deposition, which can occur during both wet and dry periods.

Our understanding of the details of the processes of transformation and deposition is not complete, but sufficient knowledge is now available that the broad outlines can be described with confidence. This is done in the succeeding sections of this chapter, with the focus on the major pollutants SO_2, NO and NO_2. The interplay of the various processes affecting emitted pollutants is depicted in Figure 1 and the important factors governing these processes are summarized in Figure 2. The complex atmospheric pathways of pollutants are apparent from these figures. Note that while SO_2 is not emitted in appreciable quantities from BTGP, it is important to the deposition of acidic precipitation in the Greater Vancouver area.

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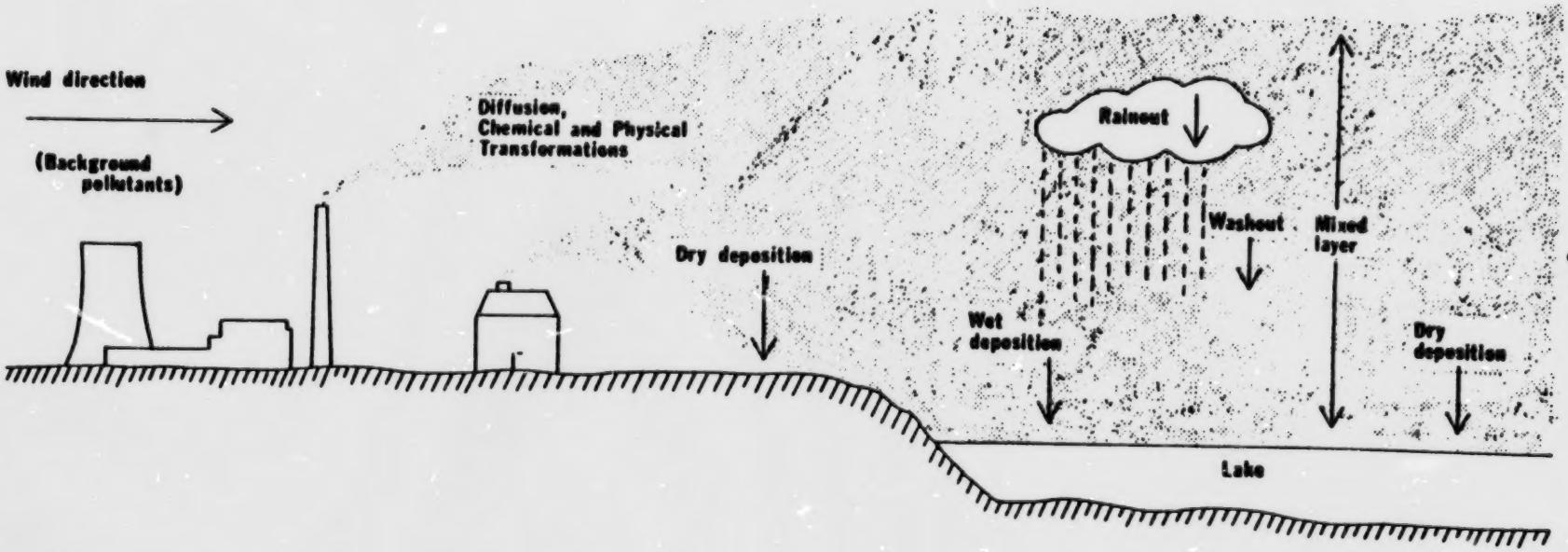
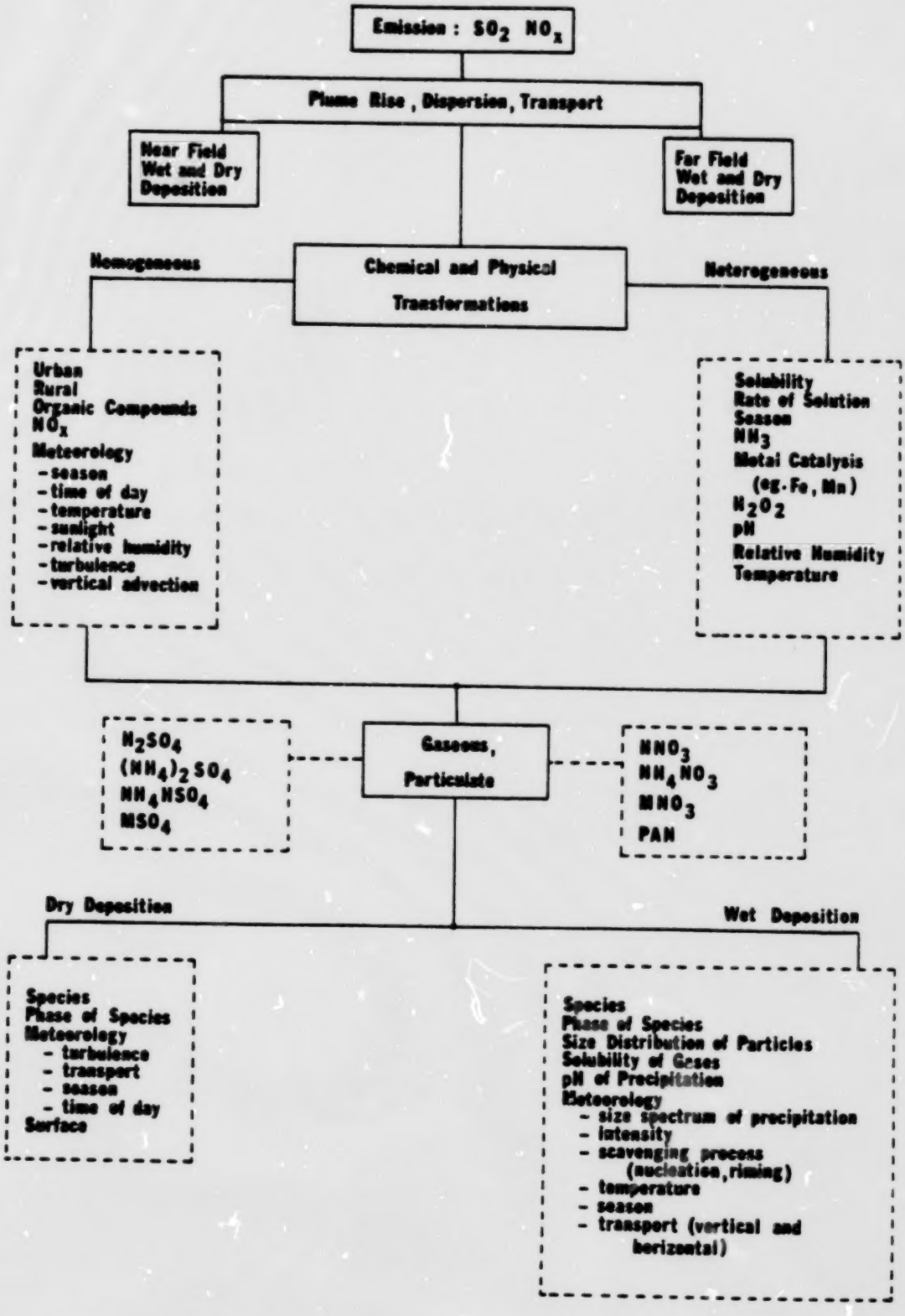


FIG. 1: THE FATE OF ATMOSPHERIC POLLUTANTS

FIG. 2 : ATMOSPHERIC PATHWAYS OF POLLUTANTS



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2.1 Chemistry of the Nitrogen Oxides

At the high temperatures prevailing in the combustion zone, nitric oxide can be formed by combination of atmospheric oxygen both with nitrogen bound in the fuel (fuel NO), and also with atmospheric nitrogen (thermal NO). Since natural gas contains essentially no nitrogen compounds, only thermal NO is formed with this fuel. Although six other oxides of nitrogen are known, they are produced in negligible quantities in combustion.

The subsequent reactions of nitric oxide may be classified as homogeneous if they take place in the gas phase, or heterogeneous if they occur in liquid water (i.e., cloud droplets, fog, rain, etc.) or on the surface of particulate matter. These reactions are discussed below, and are summarized in Figure 3.

2.1.1 Homogeneous Reactions

The gas phase chemistry of the oxides of nitrogen has been extensively reviewed by Demerjian et al (1974). More up-to-date rate constant values may be obtained from compilations, e.g. Hampson and Garvin (1977), Baulch et al (1980).

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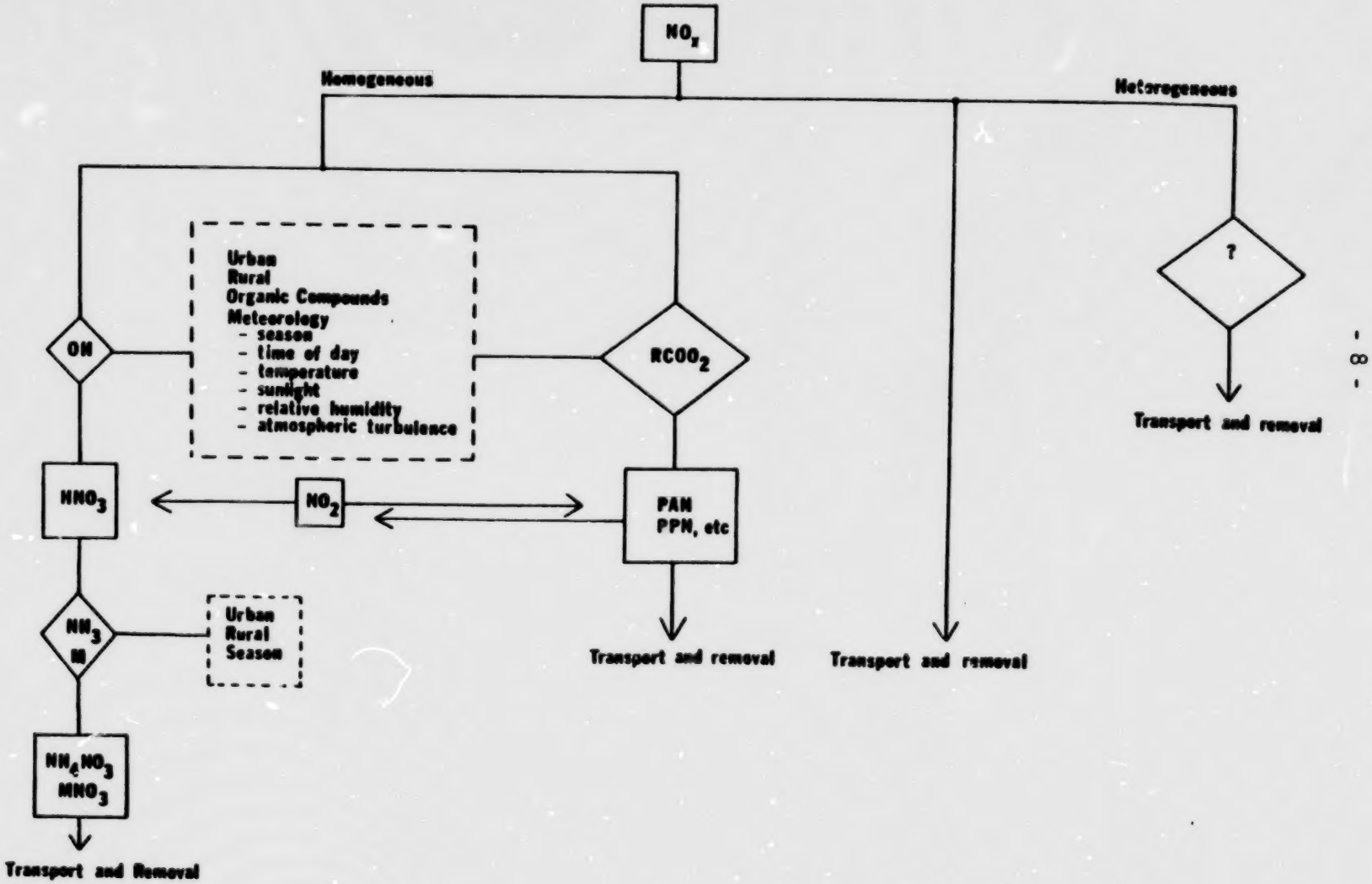
Fig.3 : ATMOSPHERIC REACTIONS OF NO_x



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The predominant oxide of nitrogen formed during combustion processes is nitric oxide, NO. As long as the concentration of this species is above about a hundred parts per million, it is oxidized at a significant rate by atmospheric oxygen.

$$2NO + O_2 \rightarrow 2NO_2$$

This reaction is responsible for the formation of a certain amount of NO_2 (typically up to 10% of the stack NO concentration) in the stack and immediately after emission, but its rate depends on the square of the nitric oxide concentration, and rapidly becomes negligible as the emitted pollutants are diluted by mixing into the atmosphere.

In a well defined plume, such as that from a power plant, the major path for oxidation of NO is by reaction with background ozone in the ambient air.

$$NO + O_3 \rightarrow NO_2 + O_2$$

The rate of this reaction is determined primarily by the rate at which the ozone mixes into the plume (Melo, 1980; Forrest et al, 1980).

Over an urban area, where there will normally be a mix of more or less aged pollutants, hydroperoxy and alkyl peroxy radicals are produced, (Demerjian et al, 1974), which are very effective in oxidizing NO as follows:

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$$NO + HO_2 \rightarrow NO_2 + OH$$

$$NO + RO_2 \rightarrow NO_2 + RO$$

These reactions are relatively fast, and, during daylight hours, most of the NO will normally be converted to NO_2 in a few hours. Since the production of the free radical species is photochemical, their concentrations tend to drop to low values at night, removing their contribution to the rate.

During the day, photolysis of NO_2 also occurs

$$NO_2 + h\nu \rightarrow NO + O$$

with subsequent formation of ozone

$$O + O_2 (+M) \rightarrow O_3 (+M)$$

where M denotes some third molecule. In the atmosphere M will normally be N_2 or O_2.

The main oxidation reactions of NO_2 involve formation of nitric acid, by:

$$NO_2 + OH \rightarrow HNO_3$$

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and formation of peroxyacetyl nitrate (PAN) by:

$$CH_3COO_2 + NO_2 \rightarrow PAN$$

The relative probabilities of these two channels depend mainly on the concentration of hydrocarbons (Spicer, 1979a). The reason is that the concentration of hydroxyl radicals appears to be almost independent of the extent of pollution of the air mass. Direct measurements of OH radical concentrations have recently become possible (Campbell et al, 1979; Perner et al, 1976; Ehhalt et al, 1981; Killinger and Wang, 1977; Wang et al, 1974, 1975 and 1981; Davis et al, 1976, 1979a, 1979b, 1981), and the reported results indicate a difference of only about 10% between the concentrations of OH in urban and rural areas. The concentration does depend strongly on solar intensity, but so do the concentrations of the free radical species involved in the formation of PAN, thus leaving the concentration of hydrocarbons, the precursors to peroxyacetyl radicals, as the major determining factor in the ratio of HNO_3 to PAN.

The final ratio of organic to inorganic nitrates depends also on temperature, since PAN is thermally dissociated at a rate which is highly temperature dependent, becoming rapid above about 25°C (Hendry and Kenley, 1979). The NO_2 thus created has a finite probability of being converted to nitric acid, rather than back to PAN, and the net effect is that PAN tends to be recycled through NO_2 to HNO_3.

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Again, these reactions are relatively rapid. The rate constant for the reaction between OH and NO_2 (Anastasi et al, 1976) is 1.1×10^{-11} $cm^3 molecule^{-1} s^{-1}$, which, when combined with a typical OH concentration of 5×10^6 cm^{-3}, yields a conversion rate of 19% hr^{-1}, in good agreement with the experimental results of Spicer (1979a).

2.1.2. Heterogeneous Reactions

Apart from the reactive dissolution of NO_2:

$$2NO_2 + H_2O \rightarrow HNO_3 + HNO_2$$

(Orel and Seinfeld, 1977; Middleton and Kiang, 1979; Lee and Schwartz, 1981), almost nothing is known about the heterogeneous reactions of the nitrogen oxides, although some reactions must occur.

However, because the homogeneous reactions of NO and NO_2 are so rapid, it is probably often true that the heterogeneous reactions are not competitive, particularly under day time conditions.

2.2 Chemistry of Sulphur Dioxide

Sulphur dioxide is formed by the combustion of sulphur contained in the fuel, and is thus not a significant species in the emissions from a gas-fired power plant. However, there are significant

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sources in the Greater Vancouver area, so that sulphur dioxide and its transformation products must be included when considering impacts on air quality and the acidity of precipitation. The gas phase and aqueous phase reactions of SO_2 are considered below, and are summarized in Figure 4.

2.2.1 Homogeneous Reactions

The homogeneous, gas phase reactions of sulphur dioxide have been extensively studied and are reasonably well understood (Calvert, 1981). The subject has been reviewed by Calvert et al (1978) and the reactions and their rates have been analyzed by Möller (1980), who has indicated the most important paths.

During daylight hours, free radical reactions predominate. The major role of the hydroxyl radical is now well established, and the rate constant for its reaction with SO_2 is known (see, for example, Möller, 1980; Calvert et al, 1978; Calvert, 1981). Combination of this rate constant and the concentration of OH, measured as described in Section 2.1.1 produces the conclusion that the rate of SO_2 oxidation by this path can be up to about 3.6% hr^{-1}.

Other important homogeneous reactions of SO_2 are with the species which may be summarized as RO_2, particularly alkyl peroxy radicals and Criegee intermediates resulting from the reaction between ozone and alkenes. A quantitative evaluation of the rate constants for

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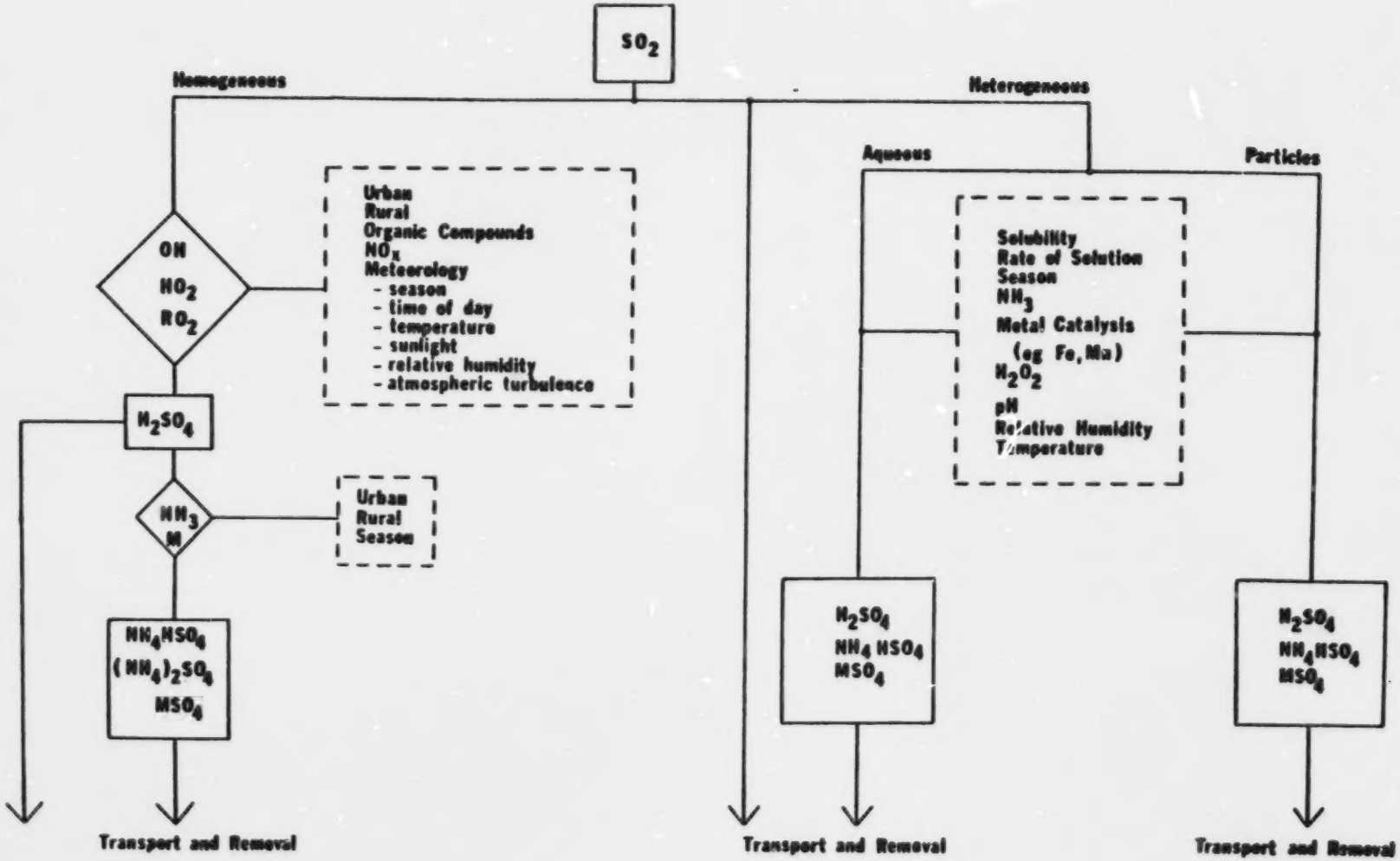
Fig. 4 : ATMOSPHERIC REACTIONS OF SO_2



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these systems is not yet available (the mechanisms are complex and subject to competition from other species such as NO) but it us believed that they are capable of making a significant contribution to the overall rate (Calvert, 1981).

At night, the concentrations of photochemically produced species drops to very low levels (for example measured concentrations of OH are 10 to 50 times lower than daytime values). However, the Criegee intermediates will persist as long as ozone and olefins are present.

It can thus be seen that, on the basis of the known rate constants, the homogeneous oxidation of SO_2 is somewhat slower than that of NO_2. This is in agreement with the results of field studies, which found that the factor probably lies in the range of three to ten (Forrest, Garber and Newman, 1980; Richards et al, 1980). It should be noted that the field measurements probably also include heterogeneous reactions.

2.2.2 Heterogeneous Reactions

Sulphur dioxide is very soluble in water (more than 20 g per 100 g water at 0°C), so aqueous phase reactions of SO_2 can be important, e.g., in clouds, fog, aqueous films on airborne particulate matter, etc. Since a significant fraction of cloud droplets are formed by conden-

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sation on solid nuclei, catalytic effects are possible in clouds as well as on aerosol particles.

Martin and Damschen (1981) have studied the solution chemistry of SO_2 and find that oxidation is fast by hydrogen peroxide but somewhat slower by ozone. The nitrite ion was also found to be capable of oxidizing SO_2.

Sulphur dioxide is oxidized by atmospheric oxygen in the presence of catalysts. Martin (1981) has indicated that iron and manganese are catalytically active, and that they are synergistic in combination, but that various forms of vanadium, copper, cobalt and lead do not catalyse the reaction.

The rates of the liquid phase reaction will be highly variable, since they depend strongly on a number of factors, such as temperature, the concentrations of catalysts, reagents, SO_2 and cloud or rain water, as well as on the acidity of the water (which affects the solubility of SO_2, and also appears to catalyse the reaction with hydrogen peroxide). To underline the possible importance of these reactions, Martin has calculated from his results that in a cloud with 1 ppb of H_2O_2 and 0.3 ml water per cubic metre, the rate of sulphate formation could be as high as 3% per minute. This is very much higher than has ever been observed in field studies, presumably because the rate very rapidly becomes limited by local depletions of reagent.

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In fact, oxidation rates obtained in the field generally lie in the range 1 to 6% hr^{-1} (Alkezweeny, 1980; Newman, 1980) suggesting that the contributions from homogeneous and heterogeneous pathways are often comparable.

2.3 Neutralization of Acids

The acids HNO_3 and H_2SO_4 produced by the oxidation reactions described above are susceptible to neutralization by other species present in the atmosphere. Chief among these are ammonia, and in some regions, calcareous dusts. The products of these neutralization reactions are largely in particulate form, though ammonium nitrate has a significant vapour pressure, and will therefore also be found in the gas phase. This has significance to deposition processes for which the efficiencies differ for gases and particles.

Ammonia is a natural component of the earth's atmosphere, being present in rather varying concentrations, which average at about 6 ppb. The source is chiefly bacterial action in soils (McConnell, 1973) and it is also produced anthropogenically (man made).

The gas phase reaction between ammonia and strong acids such as H_2SO_4 and HNO_3 is very rapid, leading to neutralization of the acids. The extent of neutralization depends on the relative concentrations of the acids and the ammonia, and also on how rapidly atmospheric mixing

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will restore local depletions of ammonia. As a further factor, the reaction between ammonia and nitric acid does not go to completion, and an equilibrium state is reached:

$$NH_3 + HNO_3 \rightleftharpoons NH_4NO_3$$

where free nitric acid remains in the gas phase. The position of the equilibrium is highly temperature dependent, with the amount of free acid increasing with increasing temperature (Stelson et al, 1979; Tang, 1980).

Thus, although the potential exists for the neutralization of free acids, it appears that it may not go to completion. For example, an analysis of the MAP3S rain chemistry measurements (Baker et al, 1981) indicates a strong correlation between acidity on the one hand and sulphate and nitrate concentrations on the other. This indicates the coexistence of H^+ and SO_4^{2-} and H^+ and NO_3^-, i.e., the existence of free nitric acid, and, at least, acidic bisulphate salts, if not free sulphuric acid. Gas phase nitric acid concentrations of the order of parts per billion are routinely measured in the atmosphere (Spicer, 1979b).

Stevens et al (1980) characterized the atmospheric aerosol in the Great Smoky Mountains and found that 61% of the mass of the fine fraction was comprised of sulphates. The average composition of these

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sulphates corresponded to ammonium bisulphate, which is acidic. Thus, the effect of acidic secondary pollutants resulting from BTGP emissions will be reduced, but not eliminated, by neutralization in the atmosphere.

2.4 Chemistry of the Nitrogen Oxides in the Vancouver Area

The meteorological conditions which are expected to give rise to maximum impacts by BTGP emissions on the Greater Vancouver area correspond to stagnation, land breeze-sea breeze episodes, arctic air outbreaks and morning fumigation episodes, (see Section 3.2.2). In these circumstances the BTGP plume cannot be considered in isolation, but generally becomes intermingled with oil refinery emissions and the Vancouver urban plume. This means that several free radical species will, as discussed in Section 2.1 and 2.2, generally be available to effect the transformation of NO to NO_2, and subsequently to nitric acid and peroxyacetyl nitrate. If the plume is emitted into relatively clean air, with no urban or local source influence, the background concentrations of free radical species other than hydroxyl will be correspondingly reduced, with a resulting reduction in the rate of oxidation of NO.

It is important to consider in some detail the conversion of NO to NO_2 in the BTGP plume, since NO_2 is a pollutant of concern in its

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own right, as well as being an intermediate stage in the conversion of nitric oxide to nitric acid. Details of oxidation mechanisms as they apply to these situations are given in the following section.

2.4.1 Conversion of NO to NO_2

As discussed above, conversion of NO to NO_2 in the BTGP plume will take place primarily via:

$$NO + O_3 \rightarrow NO_2 + O_2 \quad (1)$$

$$NO + HO_2 \rightarrow NO_2 + OH \quad (2)$$

and $$NO + RO_2 \rightarrow NO_2 + RO \quad (3)$$

while, during daylight hours, photolysis of NO_2 regenerates NO and O_3:

$$NO_2 + h\nu \rightarrow NO + O \quad (4)$$

$$O + O_2 + M \rightarrow O_3 + M \quad (5)$$

Clearly, the extent to which NO has been converted to NO_2 at any given time is a function of many parameters, including:

- Solar intensity
- Background ozone concentration
- Extent of mixing of background air into the plume
- The age of the polluted air mass into which the plume is embedded since this affects the concentrations of HO_2 and RO_2 in the air mass.

To be rigorous, a computer simulation is required to estimate the extent of conversion, using a full chemical reaction scheme, and allowing for plume dilution. This is a costly and time-consuming procedure, and to avoid its use several approximate approaches have been proposed, of which the following three seem to be the best (Cole and Summerhays, 1979):

i) Ozone limiting

This approach assumes that only ozone is effective in oxidizing NO to NO_2, and recognizes that the concentration of NO_2 which is produced may be limited by the ambient ozone concentration. That is, the concentration of NO_2 produced cannot exceed the concentration of ambient ozone. Then,

$$[NO_2] = 0.1\,[NO]_o + \text{Min}\,(0.9\,[NO]_o,\ [O_3]) \qquad (6)$$

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Where the operator "Min" indicates the smaller of the two quantities within parentheses, and the term 0.1 $[NO]_o$ allows for the initial conversion by:

$$2NO + O_2 \rightarrow 2NO_2 \qquad (7)$$

before plume dilution renders the rate of this reaction negligible.

Estimation of NO_2 in this way neglects the effect of free radicals on NO, and the photolysis and other reactions of NO_2. These are factors of potential importance in the case of the BTGP Plume and affect the NO_2 concentration in opposite directions. It is therefore not possible to say, in general, whether the concentration will be under- or over-estimated. The validity of this approach also depends on complete mixing of the plume with background air.

ii) Photostationary State

Ozone is highly reactive, and it may in certain circumstances be possible to assume that it attains a stationary state, where its rate of formation by reactions 4 and 5 is balanced by its rate of removal by reaction 1. Reaction 5 is very much

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faster than reaction 4, which is thus rate determining for ozone production. The steady state condition is therefore:

$$k_1 [NO] [O_3] = k_4 [NO_2]$$

$$[NO] / [NO_2] = (k_4) / (k_1[O_3])$$

Rearrangement of this expression yields:

$$\frac{[NO_2]}{[NO_x]} = \frac{k_1[O_3]}{k_4 + k_1[O_3]} \qquad (I)$$

Rate constant k_1 is known (Baulch et al, 1980), while the photolysis rate constant k_4 has been calculated as a function of latitude and solar zenith angle (Schere and Demerjian, 1977), so that the ratio of the NO_2 and NO_x concentrations can be calculated if the appropriate (i.e., in plume) ozone concentration is known.

However, care must be taken to ensure that the stationary state assumption is actually valid before attempting to make use of Equation (I). In particular, a suitable time must elapse before the stationary state can be reached, and this time is lengthened if complete mixing is not a rapid process.

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Further, it is possible that the stationary state assumption is not applicable if other reactions, such as those with OH, RO_2 and HO_2, cause rapid removal of NO and NO_2. It therefore seems that use of Equation (I) can provide at best a rough guide to the ratio $[NO_2]/[NO_x]$.

iii) Empirical Conversion Curves

To circumvent the uncertainties inherent in attempts to estimate NO_2 concentrations by the methods described above, it is probably preferable to make use of empirical curves obtained by measuring NO and NO_2 concentrations in the plume and plotting either $[NO_2]$ or the ratio $[NO_2]$ / $[NO_x]$ against a suitable parameter, which is usually time or downwind distance (Cole and Summerhays, 1979).

It is interesting to note that the photostationary state assumption, Equation (I) predicts that $[NO_2]$ is a linear function of $[NO_x]$, with the slope being given by:

$$S = \frac{k_1 \ [O_3]}{k_4 + k_1 \ [O_3]}$$

The value of S must therefore lie between 0 and 1.

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Care must be taken to apply such empirical data under conditions which match closely those which prevailed when the data were taken.

The conclusion is that the best estimates of NO to NO_2 conversion, as required for assessing impacts of BTGP on local air quality, are to be obtained from empirical curves of the type described above. The limits of validity, and uncertainties, associated with these curves should be borne in mind in ensuring that conservative estimates of NO_2 concentration are obtained.

2.4.2 Further Oxidation of NO_2

It has been found that the rate constant for the conversion of NO to NO_2 during the day lies in the range of about 0.5 to 1.0 hr^{-1} (Melo, 1980), with the faster rates being appropriate to the case of plumes emitted into aged masses of polluted air. The subsequent reactions of NO_2, discussed in Section 2.1, are generally somewhat slower, with the typical rate of disappearance being about 15% per hour (Spicer, 1979a).

This rate relates to formation of nitric acid plus peroxyacetyl nitrate. Both can be considered together in considering

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deposition of acids since it is believed that PAN is hydrolysed in aqueous solution to produce nitric acid (Huebert, 1981).

It follows that several hours must elapse before there will be significant production of the species which can cause acidification of precipitation. An estimate of the time taken can be obtained by assuming that the process can be approximated by the first order scheme:

$$NO \xrightarrow{k_1} NO_2 \xrightarrow{k_2} HNO_3$$

for which the rate expression can be integrated analytically. If k_1 is assumed to lie in the range 0.25 to 1.0 hr^{-1}, with k_2 having the value 0.15 hr, in accordance with the discussion above, the fraction of initial NO which has been converted to nitric acid is as given in Table 1 as a function of reaction time. The rate constant values assumed above are based on those quoted by Melo (1980). the lower limit for k of 0.25 hr^{-1} was chosen to reflect the faster conversion rate which might be expected in the BTGP plume because of the urban location.

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Table 1

Fraction of NO Converted to HNO_3

Time/Hours	Fraction
1	0.03 - 0.06
2	0.08 - 0.18
3	0.15 - 0.30
4	0.23 - 0.40
5	0.30 - 0.50

The range in the fraction corresponds to the range assumed for the rate constant for the oxidation of NO to NO_2 (0.25 - 1.00 hr^{-1}). It can be seen that travel time is very important in assessing the impact of source emissions. This fact takes on great importance later in the report.

To allow a semi-quantitative assessment of the additional effect of BTGP emissions on acidic deposition in the Vancouver area, a simple model was set up, involving emissions, physical and chemical transformations and deposition of pollutants. The model is described in Appendix 2 to this report, and its use, and the conclusions derived from it are discussed in Chapter 4. One of the simplifications incorporated in the model is the assumption that the formation of HNO_3 from NO occurs in a single step, which can be described by first order kinetics. The justification for this assumption is discussed below.

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For the hypothetical case

$$NO \overset{k}{\updownarrow} HNO_3$$

the kinetic rate law integrates to

$$[HNO_3] = [NO]_0 \, [1 - \exp(-kt)]$$

where:

$[HNO_3]$ = HNO_3 concentration at time t

$[NO]_0$ = initial NO concentration.

Rearrangement gives

$$- \ln \left[1 - \frac{[HNO_3]}{[NO]_0}\right] = kt$$

which indicates that a plot of $- \ln \left[1 - \frac{[HNO_3]}{[NO]_0}\right]$ against t should be a straight line of slope equal to the rate constant, k.

Such a plot is shown in Figure 5 for the two extreme cases of HNO_3 formation presented in Table 1. The good fit to a straight line for both cases confirms that the assumption of single step reaction is valid after an initial period of about 40 to 60 minutes, during which essentially no nitric acid is formed.

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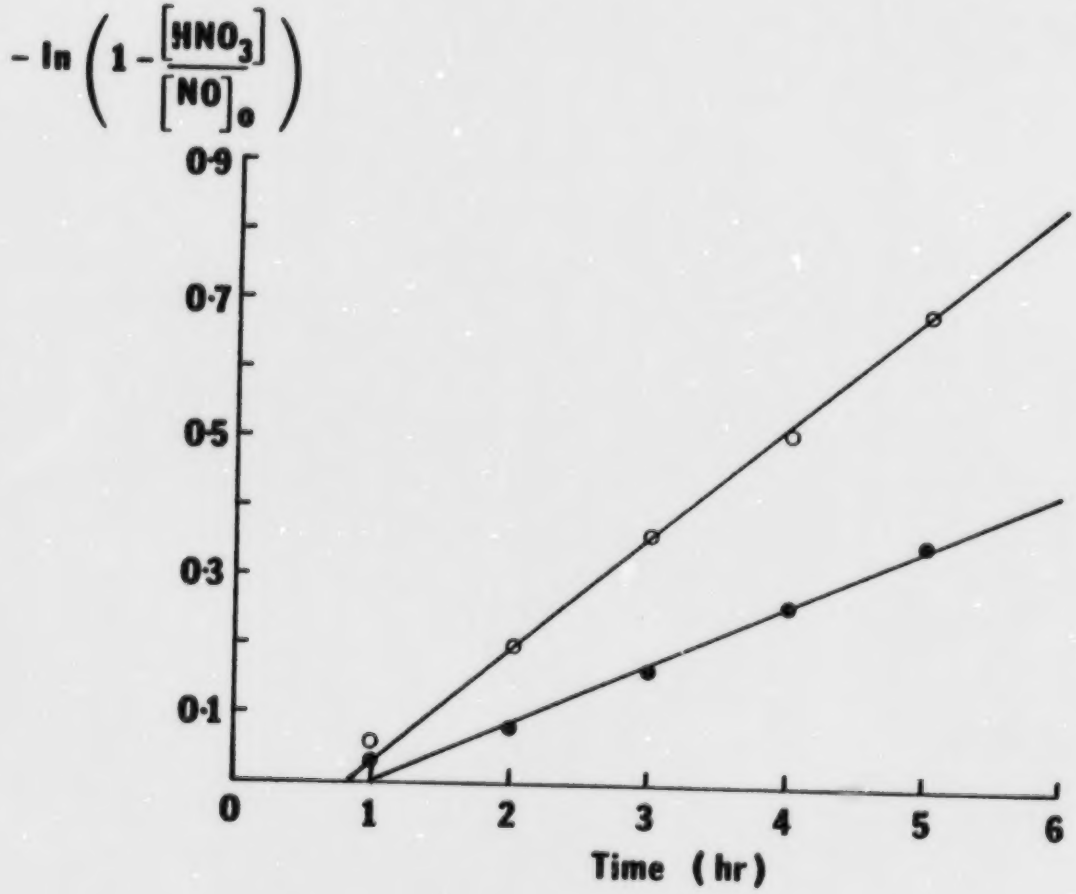
Fig. 5 : Kinetic Plot for HNO_3 Production



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This initial delay is known as an induction period, and it arises because the reaction actually proceeds through an intermediate, which must first be formed in appreciable amounts before any significant concentration of the final product can be formed.

The implications of the results expressed in Figure 5 to the operation of the BTGP are very important, since they indicate that for the first 40 to 60 minutes after emission, essentially no acidifying species are formed in the plume.

It should be reiterated that the conversion rates presented and discussed above relate to daylight hours only. the night time rates will generally be significantly lower.

2.5 Atmospheric Deposition

Atmospheric deposition may be divided into wet deposition, which results from the entrainment of pollutants into precipitation, and dry deposition, which describes the processes of absorption of gaseous and particulate material by land and water surfaces.

2.5.1 Wet Deposition

Wet deposition may be subdivided into rainout and washout, depending respectively on whether the interaction with the pollutant occurs in the cloud or below the cloud.

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In both processes particles are captured by impaction and interception, and Brownian motion. Various electrical and phoretic mechanisms are also involved. In addition, hygroscopic particles can act as condensation nuclei in clouds, and the subsequent rainout of this material results in particularly efficient removal of sulphate and nitrate particles from the atmosphere.

In the case of gases, the processes of diffusion to the droplet, followed by absorption are important, particularly for sulphur dioxide. Chemical transformation is then likely to occur (see Section 2.2.2).

Wet deposition is often parameterized in terms of a scavenging coefficient, though the physical and chemical processes involved are known to be complex (Barrie, 1981). The scavenging coefficient, λ , is defined by:

$$C_t = C_o \exp(-\lambda t)$$

where

C_t = concentration of the material at time t

C_o = its initial concentration

An alternative parameterization is in terms of the washout ratio, which is defined as the concentration of contaminant in precipitation divided by its concentration in air (usually at the surface level), so that:

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$$W = x/c$$

where

x = concentration of contaminant

c = its concentration in air

W = washout ratio

The scavenging coefficient and washout ratio are related. For example, if a pollutant of uniform concentration in a column of air of height h is being washed out by precipitation of intensity J, it can be shown that:

$$\lambda = WJ/h$$

Therefore, the following discussion will be focussed on the scavenging coefficient, with the understanding that the conclusions apply also to the washout ratio.

The scavenging coefficient depends on a number of meteorological parameters, as well as on the properties of the material being scavenged. Specifically, it is generally agreed that precipitation type and rate are important, as is the size spectrum of the precipitation. For particles, the size distribution and solubility are important, as is solubility for gases, including its possible dependence on the temperature and acidity of the precipitation.

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Theoretical understanding of wet deposition has not yet advanced to the point where scavenging coefficients can be calculated from the known chemical and physical properties of the pollutant and the meteorological parameters, and, as a result, it is necessary to rely on experimental results.

Most of the experiments which have been carried out to date have focussed on sulphur dioxide, and there are few data available on sulphates or the nitrogen compounds. Measured values of λ for SO_2 exhibit large variability, which may reflect, in part at least, the strong dependence of SO_2 solubility on solution pH. Inspection of the available data (summarized by Lusis and Shenfeld, 1981) suggests that λ for SO_2, sulphates and nitrates is of the order of 1-5 x 10^{-5} sec^{-1}, while λ for NO_2 is approximately 1/4 as large. On grounds of solubility, λ for NO should be approximately a further order of magnitude smaller.

Wet deposition of pollutants is an important process in the Greater Vancouver area; however, the primary pollutants emitted by the BTGP are not efficiently scavenged.

2.5.2 Dry Deposition

In dry deposition the mechanisms bringing pollutants to the earth's surface include impaction, gravitational settling and Brownian

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motion for particles, and molecular diffusion and absorption for gases. However, for both particles and gases, turbulent diffusion within the planetary boundary layer is probably the predominant factor.

Dry deposition is usually discussed in terms of the deposition velocity, v_d, defined by:

$$F = v_d C$$

where F = flux to the ground of the species

and C = its ambient concentration.

Because of the role played by turbulent diffusion in the deposition process, it is clear that v_d will depend on meteorological parameters such as atmospheric stability and wind speed, and also on surface characteristics, particularly roughness. The deposition velocity also depends on the probability that the particle or gas molecule sticks once it impacts on the surface, which is determined primarily by the condition of the surface (particularly moisture content) and the solubility of the pollutants.

Most of the experimental results obtained to date relate to sulphur dioxide. For a wide range of surfaces, including vegetation, water and soils, v_d has been found to lie in the range 0.4 - 0.8 cm sec^{-1}. For snow surfaces v_d is somewhat smaller (about 0.1 - 0.2 cm

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sec^{-1}), and it appears that the deposition velocities for SO_2, NO_2 and NO are approximately in the ratio 1.0 to 0.5 to 0.1 (Judeikis and Wren, 1978; Sehmel, 1980).

The dry deposition of sulphate particles is not well understood. Lusis and Shenfeld (1981), summarized the data and found a large number of low values, including some which were negative, but other workers have reported velocities comparable to those for SO_2 (Sheih et al, 1979). No data are currently available for particulate nitrates or nitric acid. For gaseous nitric acid, a reasonable assumption is that v_d is approximately the same as that for SO_2, since both species are very soluble in water. The rate at which the gaseous molecules are brought to the surface will be approximately the same for both species, so that the relative values of v_d will depend on the relative probabilities of sticking once they strike the surface. This in turn is largely dominated by uptake into fluids, e.g., in the stomatal openings of plants, etc. The lack of information on particulate nitrates constitutes a serious data gap.

2.6 Meteorology

All of the chemical and physical processes taking place in the atmosphere are influenced by meteorology. It is beyond the scope of this report to discuss these influences in detail, but some of the more important ones are summarized below:

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- Large scale pressure systems determine the large-scale flow circulation which, in turn, determines the long range transport of pollutants.

- The physical and chemical conversion processes of pollutants are determined by the lifetime of those pollutants in the atmosphere. Such factors as frequency and type of precipitation, solar radiation, transport velocity and direction, mixing heights, temperatures and air mass trajectories can have a strong influence on this lifetime.

- Dry deposition rates are determined by both large-scale and small-scale atmospheric turbulence. The former is responsible for diffusing pollutants to the surface and the latter for transporting the pollutants to the actual pollutant sink.

- Wet deposition rates are determined by the type and frequency of precipitation, the characteristics of precipitation fields in receptor areas, the types of storms responsible for the precipitation (e.g., convective, frontal) and the flow dynamics of the storms.

- Local meteorological effects such as stagnation or land breeze-sea breeze episodes are often responsible for high air pollution and dry deposition conditions. In such cases local

pollutant emissions are not transported out of a region and may accumulate for several days. This may result in high, persistent pollutant concentrations, and providing ideal conditions for chemical and physical transformation of the primary pollutants. However, GVRD monitor data indicate that accumulation of NO_2 and NO_x does not necessarily occur during stagnation episodes.

Most meteorological factors which affect wet and dry deposition are complex and variable. Chapter 3 includes a description of those factors which are most important to the present assessment.

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3. BACKGROUND INFORMATION APPLICABLE TO THE BTGP

The following sections review data available from previous studies and reports which are of use in assessing the potential impact of the operation of the BTGP on the deposition of acidic compounds. Included are data on the emission of acid precursors (SO_2 and NO_x) within the Greater Vancouver area, prevailing meteorological conditions, available precipitation chemistry and air quality data, and lake chemistry data.

3.1 Emissions

The pollutant emissions of primary concern here are NO_x (NO_2 and NO) and SO_2 since both are acid compound precursors. However, also of importance are organic compounds since these pollutants, in combination with NO_x and sunlight, react chemically in the atmosphere to form photochemical smogs which are characterized by a strong oxidizing capability. Thus, in the presence of photochemical smog, the conversion rate of both SO_2 and NO_x to acidic compounds is accelerated.

3.1.1 Greater Vancouver Area Emissions

Emission inventory data for the Greater Vancouver area are given in Table 2. Total emissions for 1976 are given for SO_2, NO_x (expressed as NO_2) and volatile organic compounds. These data were

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Table 2

GVRD Emissions of SO_2, NO_x and
Volatile Organic Compounds (VOC) (1976)
(tonne/yr)

	SO_2	NO_x	VOC
Application of Surface Coatings	-	-	7,739
Diesel and gasoline marketing	-	-	11,063
Diesel powered vehicles	2,077	14,200	1,400
Dry cleaning	-	-	1,504
Stationary combustion sources (Space heating, power generation, etc.)	24,767	13,480	561
Gasoline powered vehicles	1,279	24,000	40,300
Non-highway use of gasoline	57	1,219	2,161
Off-highway mobile sources	1,823	7,964	2,492
Petroleum refining	17,100	2,309	11,745
Incineration	44	48	15
Structural fires	-	-	342
Tire wear	-	-	32
Total	47,147	63,220	79,354
BTGP	-	3,660	-

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obtained mainly from the National Emission Inventory System (NEIS) maintained by the Environmental Protection Service of Environment Canada. The exception is the data for mobile sources, for which data from the Mobile Sources Division of the Air Pollution Control Directorate were preferred (Concord Scientific Corporation, 1982).

The total SO_2 emissions in the Greater Vancouver airshed for 1976 were estimated to be about 47.1×10^6 kg/yr. The largest source category was space heating/power generation/waste incineration at about 53% or 24.8×10^6 kg/yr. Industrial emissions of SO_2 were also large at 38% of the total or 17.1×10^6 kg/yr, with virtually all of this derived from petroleum refining. Although the geographic distribution of these emissions is not readily available, it is significant that the refinery SO_2 emissions (36% of total) are all located along Burrard Inlet, close to the BTGP.

Greater Vancouver area emissions of NO_x showed a marked difference in category distribution from SO_2. Total emissions for the area in 1976 were estimated to be 63.2×10^6 kg/yr. Of this amount, gasoline vehicle emissions dominated at 38% or 24.0×10^6 kg/yr, with the total for all mobile sources amounting to 70% (46×10^6 kg/yr). Space heating/power generation/waste incineration were also significant at 21% of the total or 13.5×10^6 kg/yr. Industrial sources of NO_x account for only about 4 % or 2.3×10^6 kg/yr. However, of this category, almost all emissions were due to oil refining operations which were located

relatively near to the BTGP along Burrard Inlet. Motor vehicle emissions were distributed over the urban area consistent with traffic patterns.

Total emission of organic compounds was estimated to be 79.4 x 10^6 kg/yr, arising primarily from petroleum refining, gasoline marketing and mobile sources. The respective contributions from these sectors were petroleum refining 11.7 x 10^6 kg/yr (14 %) and transportation sources 44.2 x 10^6 kg/yr (56 %). It should be noted that the refinery emissions are all located relatively close to BTGP.

3.1.2 Burrard Thermal Generating Plant Emissions

A source testing programme was carried out in order to characterize emissions from the BTGP under maximum load conditions (B.C. Hydro (1982). The results indicated a maximum flue gas concentration of 145 ppm NO_x expressed as NO_2 (determined at maximum load). The in-stack NO to NO_x ratio was assumed to be 0.95, based on tests made at other plants. Under natural gas firing no significant quantities of SO_2 or hydrocarbons are emitted.

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3.1.3 Comparison of the BTGP Emissions with Total Area Emission Inventory

In order to place emissions from the BTGP in context with the total area emissions, a comparison of emissions is useful. However, in assessing potential incremental impacts, several other factors must also be examined. These include factors such as geographic distribution of emissions, local meteorological conditions and other pollutants present in the background air mass.

It should be noted that the BTGP emission figures given below assume a 65.7 % capacity factor which is equivalent to continuous operation of the plant at its historical normal maximum load. The assumption of continuous operation for BTGP will result in higher than expected emissions of NO_x since the facility would normally shut down when hydro power is available. If other power outputs are used the emissions would need to be scaled accordingly.

The Greater Vancouver area NO_x emissions for 1976 were estimated as 63.2×10^6 kg/yr. The emissions from the BTGP of 3.7×10^6 kg/yr represent only about 6 % of this figure. Emissions of NO_x within Burrard Inlet totalled about 2.3×10^6 kg/yr from oil refineries, to which must be added an unknown amount from transportation sources. The BTGP would not contribute additional SO_2 or hydrocarbons to the area emissions.

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A potentially more useful figure for comparison is to express emission of NO_x and SO_2 in terms of acid equivalents. The realization of acid formation from SO_2 and NO_x of course depends on the emission source distribution and the transport, diffusion, conversion and deposition processes which are discussed in Chapter 2 and Chapter 4. Therefore the comparison of emissions in terms of acid potential should be treated with caution.

Table 3 is a summary of SO_2 and NO_x emission data for 1976 in the GVRD expressed as equivalent weight of acidic hydrogen ions. Note that each molecule of SO_2 produces two acidic hydrogen ions, while NO_x produces only one.

Area SO_2 emissions lead to a total potential acid burden of about 1.47×10^6 kg/yr. Of this, SO_2 emission within Burrard Inlet due to oil refinery operations amounted to 0.53×10^6 kg/yr. Area NO_x emissions translate into a total potential acid burden of about 1.38×10^6 kg/yr. Of this total, about 0.05×10^6 kg/yr was attributable to oil refinery emissions within Burrard Inlet. Hence the total potential acid equivalent emissions for the area was 2.85×10^6 kg/yr of which oil refinery emissions within the Burrard Inlet represent 0.58×10^6 kg/yr. By comparison, potential acid equivalent emissions from the BTGP are 0.083×10^6 kg/yr (14 % of refinery emissions or 3 % of area totals).

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Table 3

GVRD 1976 Emissions of SO_2 And
NO_x Expressed as Acidic
Hydrogen Ion Equivalents
(tonne/yr (% of total))

Source Sector	SO_2	NO_x	Total
Diesel powered vehicles	65 (4)	309 (22)	374 (13)
Stationary combustion sources	774 (53)	293 (21)	1067 (37)
Gasoline powered vehicles	40 (3)	522 (38)	562 (20)
Non-highway use of gasoline	2 (0)	27 (2)	29 (1)
Off-highway mobile sources	57 (4)	173 (13)	230 (8)
Petroleum refining	534 (36)	50 (4)	584 (21)
Incineration	1 (0)	1 (0)	2 (0)
TOTAL	1473 (100)	1375 (100)	2848 (100)
BTGP	-	80 (6)	80 (3)

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The following points summarize the important aspects of Greater Vancouver SO_2 and NO_x emissions as compared to maximum expected BTGP emissions of NO_x.

1. Area emissions of SO_2 are substantial, no SO_2 is emitted from the BTGP.

2. A significant fraction (36 %) of the area SO_2 emissions are from oil refineries located along Burrard Inlet close to the BTGP.

3. Area emissions of hydrocarbons are substantial and essentially no hydrocarbons are emitted from the BTGP. Hydrocarbons assist in converting SO_2 and NO_x into acidic compounds.

4. A significant fraction (15 %) of hydrocarbon emissions are emitted from oil refineries within Burrard Inlet close to the BTGP.

5. Maximum BTGP emission of NO_x amounts to about 6% of total area emission.

6. In terms of total potential acid burden for the GVRD area, SO_2 accounts for about 52 % and NO_x, 48 %.

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7. Emissions from BTGP operating continuously at normal maximum load give rise to a potential acid burden of about 3 % of the area total and 14 % of that due to oil refineries within Burrard Inlet.

3.2 Meteorological Information

The influence of meteorology on wet and dry deposition has been briefly discussed in Chapter 2. Based on that general discussion, this section describes those meteorological factors expected to influence deposition fields in the Greater Vancouver area. Background data on these factors are also presented.

The information is divided into 4 topic areas: wind fields, dispersion conditions, storm systems and precipitation fields. It should be noted that these factors are not mutually exclusive but are highly inter-related.

3.2.1 Wind Fields

The transport of pollutants in the Vancouver area is controlled by the wind fields on the macro-, meso- and micro- scales. On the macro-scale, it is the large scale circulation which transports background pollutants into the region, determines local flow, and accounts for the Pacific frontal weather systems which produce most of the local precipitation.

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On the meso-scale, factors such as the sea breeze - land breeze circulation and stagnation episodes determine the lifetime and path of pollutants in the area; over-riding air produces frontal precipitation and frontal waves with their typical flow patterns. Also, terrain effects on the meso-scale produce convergence and divergence as well as drainage and upslope flows.

On the micro-scale, small scale flow patterns near pollution sources affect the transport, dispersion and deposition of pollutants.

In order to carry out a rigorous assessment of the impact of a particular source on wet and dry deposition fields, it is desirable to assess the flow fields associated with all of these scales. However, this type of information is simply not available. In this case, information that is available is low-level wind direction and wind speed frequencies in the Burrard Inlet area. These are presented in detail by Emslie (1981).

This reference indicates that easterly winds (i.e. SE to NE sector) predominate throughout the year along Burrard Inlet. However, during the summer months, westerly flow (i.e. SW to NW sector) also becomes very frequent. Local influences such as drainage flows from the mountains and land breeze-sea breeze circulations probably account for much of this behavior.

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At Burrard Thermal Generating Plant the wind statistics reflect the same pattern with predominant southwest to northwest and northeast flow.

These wind statistics strongly suggest that for most of the year low level winds are channelled along Burrard Inlet. The implications of this are that BTGP emissions often travel to the west, where with travel times in the order of an hour, they are thoroughly imbedded in the Vancouver urban plume and have an incremental impact on the urban area. In the summer months, westerly flow also occurs fequently so that urban emissions are transported east along Burrard Inlet and BTGP emissions are emitted directly into the urban plume. The eastern Burrard Inlet area would therefore be affected by the urban emissions. The westerlies responsible for this transport are sea-breezes, and thus occur principally during the day. Easterly winds typically prevail at other times.

The relatively frequent SW component of wind at BTGP means that emissions could also travel into the mountains to the northeast of the plant with a significant frequency.

This discussion so far has focussed on annual all-hour statistics. Similar statistics are available for precipitation periods only (Emslie, 1982). Based on these statistics prevailing flow during precipitation episodes is from the northeast, east and southeast. This is discussed in detail in Section 3.2.3 Storm Systems.

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The wind statistics discussed above are useful for determining where most of the wet and dry deposition of BTGP emissions could take place. It is felt that most dry deposition will occur in the sectors to the east and west of the plant. It should be noted that these statistics refer only to low-level winds and that the BTGP emissions will be influenced by upper level winds also. Although wind shear is expected between the upper and lower level flows, it is believed that general flow patterns presented here will apply reasonably well to BTGP emissions since dry deposition takes place at ground level.

3.2.2 Dispersion Conditions

Wet and dry deposition of pollutant emissions depends not only on the direction of transport of emissions but also on the concentration and, therefore, the amount and dispersion of the emissions. For example, a plume must be dispersed to the ground level in order for dry deposition to take place. An often used method for determining dry deposition is to multiply the concentration of the pollutants at ground level by a characteristic deposition velocity (Hicks, 1981), i.e. $D = V_d C$, where D = dry deposition (mass/area-time), V_d = deposition velocity (length/time), and C = concentration (mass/volume).

There is a very little information available on the dispersion climatology of the Burrard Inlet area. That is, information such as a

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mixed layer climatology, inversion frequency analysis, plume impingement statistics and/or Pasquill stability categories are not available. Because it is not available, the approach taken in the assessment is to consider that local emissions are mixed thoroughly and relatively quickly in a mixed layer. This is oversimplified but does allow one to err on the conservative side.

In specific terms, there are four situations whereby wet and dry deposition can be severely affected by dispersion conditions. The four situations are described in detail in a B.C. Hydro report (B.C. Hydro, 1982) but have been briefly summarized (based on the B.C. Hydro report) below for reference. They are:

1. Fall Stagnation Periods - these periods are caused by the weakening of local pressure gradients due to the gradual collapse of the Pacific high pressure cell. They are characterized by light winds, strong surface-based inversions and the building of local emissions in the air mass. The frequency of these events can vary from year to year ranging from as little as no episodes to as much as one nearly continuous episode lasting from late August to early October.

2. Winter Arctic Air Outbreaks - in these situations, outbreaks of cold dry Arctic air undercut the residual Pacific air over

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the B.C. coast causing strong surface inversions. If the extent of the outbreak is large, it can eventually result in light winds and very intense inversions.

3. Spring and Summer Land Breeze - Sea Breeze Episodes - these episodes occur when the Pacific high pressure centre is located well off the B.C. coast resulting in a weak pressure gradient over the mainland. The result is an osciilating daytime and nighttime sea breeze-land breeze situation with their corresponding strong inversions. During these periods, local emissions can be carried back and forth across the mainland and may build up within the air mass. Between 20 and 40 land breeze-sea breeze episodes, lasting 3 to 11 days, occur on average per year.

4. Morning Fumigation Episodes - these occur after calm, clear nights when the nighttime radiation inversions break up and pollutants fumigate to ground level.

3.2.3 Storm Systems

The deposition of pollutants by wet mechanisms depends not only on the type and amount of pollutant but on such meteorological factors as the type of precipitation, temperature, types of storm system (frontal versus convective) and size distribution and intensity of

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precipitation. When assessing the potential effect of a pollutant source on the wet deposition field in a given area, it is desirable to assess each of these factors in concert with the frequency with which the emissions are transported in different directions. This is particularly important when certain receptors in the area are sensitive to acid deposition. For example, if emissions from a given source are seldom transported towards a poorly buffered lake during precipitation episodes, then that lake will not be affected greatly by the source.

In assessing the impact of BTGP emissions on wet deposition in the Vancouver area, the most useful information on storm characteristics is provided by Schaefer (1980, 1981) and Emslie (1982). The former references indicate that most of Vancouver's precipitation (which occurs from October to March) is caused by Pacific frontal systems which pass over the area with relatively high frequency. Preceeding the fronts are strong southeasterly winds and heavy rains. In general, in the Lower Fraser Valley, these winds are easterly rather than southeasterly.

After the frontal passage, the winds shift to the southwest and precipitation ceases. On the average, the fronts pass over the Vancouver area every few days in the winter. Summer convective showers account for very little of the local precipitation.

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Emslie (1982) provides a more quantitative analysis of flow characteristics during precipitation events through the case study of a typical precipitation episode, picked at random, and through the presentation of precipitation - only wind statistics for Vancouver Airport and Burnaby Mountain. The case study was for a Pacific frontal storm which passed over the lower mainland from noon, December 1, 1981 to noon December 2, 1981. Wind data during the event were available from Vancouver Airport (10 m MSL) and from Burnaby Mountain (465 m MSL). Prior to the start of precipitation the winds at Burnaby Mountain were from the east - southeast. During the precipitation period, the winds were predominantly from the east-northeast; with the passage of the front and the end of precipitation the wind swung through the south to the southwest. At Vancouver Airport the winds were similar to those at Burnaby Mountain prior to and during most of the precipitation. However, the post-frontal wind shift to the west was more abrupt than on the Mountain.

Further quantification of flow directions during precipitation is provided by the precipitation-only wind statistics in Emslie (1982). The statistics were generated by the Atmospheric Environment Service of Environment Canada. They include monthly wind speed and direction frequencies for Burnaby Mountain and Vancouver Airport for periods of precipitation recorded at Vancouver Airport. The results of the statistical analysis, from Emslie, are shown in Appendix 1 and are summarized in Table 4a. The table shows the highest and second highest frequencies

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of wind direction during precipitation episodes for both Burnaby Mountain and Vancouver Airport. The table indicates that the predominant wind direction during precipitation is easterly at both locations, more strikingly so at Vancouver Airport with at least 50 % of its winds from that direction every month. However, Burnaby Mountain shows a northeasterly predominance in its second-highest wind direction frequencies while Vancouver Airport shows a corresponding predominance of southeasterly winds.

The areas of highest impact of BTGP on wet deposition in the Vancouver region can be assessed from the foregoing statistics if one assumes that the BTGP plume is transported at an elevation between that of Burnaby Mountain and Vancouver Airport. This assumption is supported by plume studies carried out by B.C. Hydro (1982). It is clear from Table 4a that the predominant direction of BTGP emissions during precipitation is to the west - along Burrard Inlet and over the city of Vancouver. However, the statistics also suggest a strong influence of southeasterly and northeasterly winds depending upon the location and the elevation of the emissions. The cumulative percentage of winds from the northeast-east-southeast sector for the two anemometers is given in Table 4b and confirms the strong predominance of winds from that sector. Therefore,the greatest potential wet deposition impact of the BTGP emissions is likely in the southwest - west - northwest sector, (i.e. towards Vancouver) where they will mix with local urban emissions.

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TABLE 4a

Summary of The Highest and Second Highest Precipitation - Only Monthly Wind Direction Frequencies From Burnaby Mountain and Vancouver Airport (in Parentheses)

Direction	Month											
	J	F	M	A	M	J	J	A	S	O	N	D
N	23	-	-	-	-	-	-	-	-	-	-	-
NE	21	18	19	22	-	-	-	19	17	21	17	22
	(11)	-	-	-	-	-	-	-	-	-	-	(14)
E	-	24	25	23	31	25	27	35	33	36	29	19
	(56)	(51)	(50)	(51)	(50)	(51)	(53)	(59)	(60)	(59)	(56)	(60)
SE	-	-	-	-	-	-	-	-	-	-	-	-
	(11)	(20)	(21)	(19)	(19)	(19)	(13)	(17)	(15)	(17)	(15)	-
S	-	-	-	-	19	18	19	-	-	-	-	-

TABLE 4b

Summary of Precipitation - Only Wind Frequencies From the Southeast - East - Northeast Sector

Location	Month											
	J	F	M	A	M	J	J	A	S	O	N	D
Burnaby Mountain	44	55	59	68	53	52	52	64	65	69	62	55
Vancouver Airport	78	82	78	79	74	78	72	84	86	84	83	86

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3.2.4 Precipitation Fields in the Vancouver Area

The impact of a pollution source on wet deposition is a function of the frequency, amount and type of precipitation falling in the area. In the Vancouver region, all three of these factors are highly variable in time and space and therefore have potentially significant effects on the local wet deposition field. A brief summary of the more important characteristics of the precipitation fields is given below.

a) The amount of precipitation falling in the Vancouver area is greatest during the period from October to March (this period is referred to as winter in the balance of the discussion). This is graphically illustrated in Figure 6 which shows the 20 year average monthly precipitation rates (AES, 1981) for four locations in the Vancouver area (Vancouver Airport, IOCO Refinery, Buntzen Lake and Coquitlam Lake). It is apparent that the precipitation amounts differ quite significantly from the winter months to summer months. Table 5 shows that winter precipitation is roughly 3 times greater than summer precipitation.

Schaefer (1980) attributes this difference to the increased frequency of Pacific frontal systems passing over Vancouver in the winter. Also, in the summer, the presence of the North Pacific anticyclone tends to reduce the number of frontal storms.

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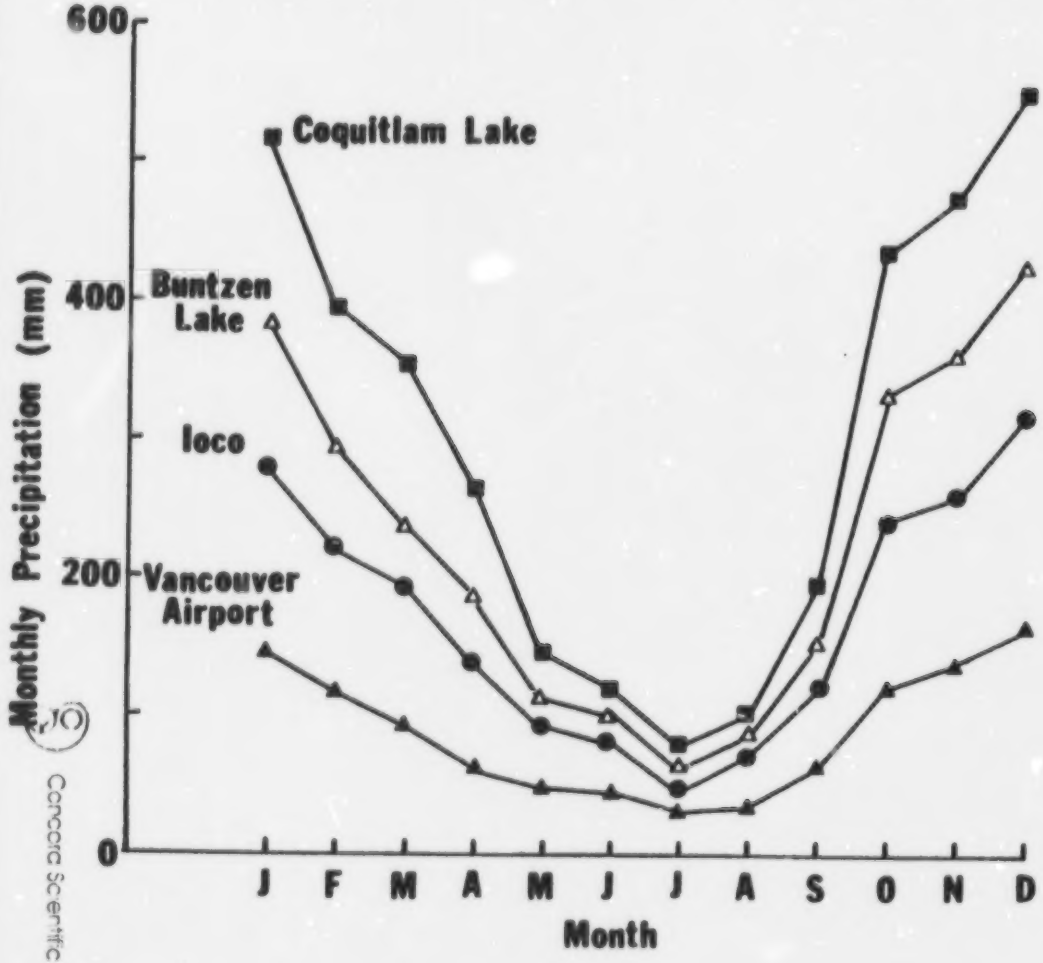
Fig. 6 : Monthly Precipitation Rates
(30 year average)

(From the Atmospheric Environment Service)



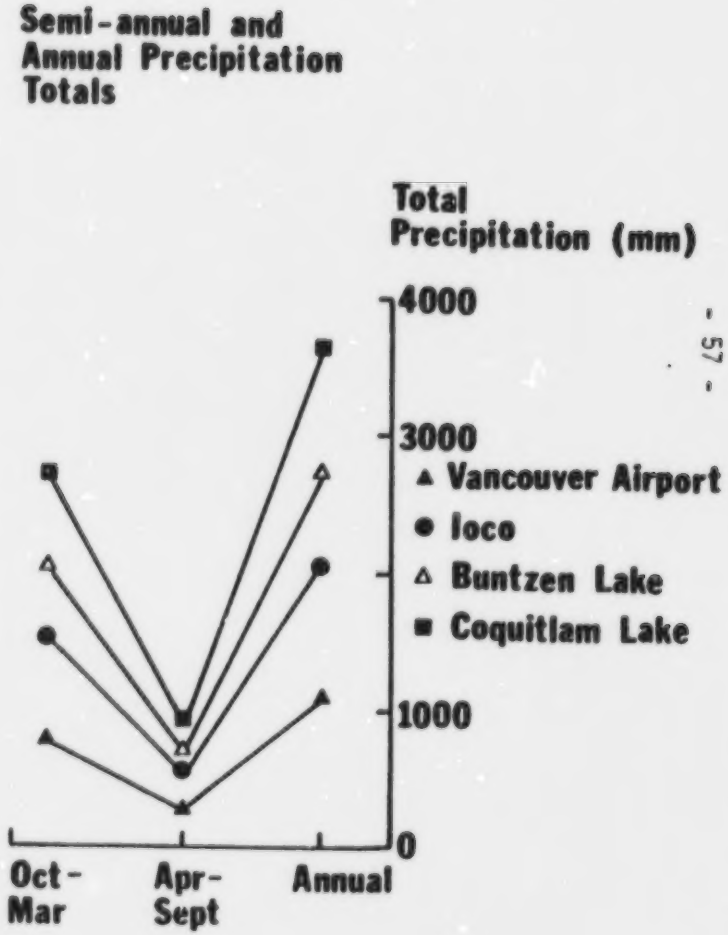


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Table 5

20 Year Average Summer/Winter Precipitation Rates

LOCATION	SUMMER (April-Sept.) TOTAL (mm)	WINTER (October-March) TOTAL (mm)	RATIO Winter/Summer
Vancouver Airport	281.7	786.4	2.79
IOCO	548.6	1511.5	2.76
Buntzen Lake	699.7	2038.8	2.91
Coquitlam Lake	903.0	2728.2	3.02

Figure 6 also points out that the maximum monthly precipitation occurs in December and the minimum in July.

b) The freqency of precipitation episodes is also greater for October to March than for April to September. This is shown in Figures 7 and 8. In Figure 7 one can see that the average number of days with precipitation greater than 0.2 mm is roughly 1.5 to 2 times greater in the winter than the summer. Again, the peak appears in December and the minimum in July. Careful inspection of Figure 7 also shows that, during the winter, precipitation occurs in over half the days of each month. This is further broken down in Figure 8 which shows the average number of hours per month with precipitation recorded at Vancouver Airport. This figure shows that precipitation occurs on average roughly 22 % of the hours in the winter months and 11 % of the hours in the summer months. The annual average is 16.6 %.

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c) The times of the day when precipitation occurs most frequently also vary with season (Wright and Trenholm, 1969). In January, for instance, the time of most frequent precipitation is from 5 to 7 am with a secondary maximum for 8 to 9 pm. The period of least frequent precipitation is during the afternoon. In July, the greatest frequency of events is in the late morning and early afternoon with the minimum in the late afternoon.

d) The extreme changes in topography around Vancouver play a very significant role in determining the amount and type of precipitation falling in the area. There are two general effects of the topography (Schaefer and Nikleva, 1973); the first is the orographic effect which results in increasing precipitation with increasing elevation; the second is the convergence effect (funnelling of air into mountain valleys). As a result, the amount of precipitation in the Vancouver area increases significantly with elevation and tends to maximize on the mountain barriers at the heads of open valleys. The effect of these influences can be seen in Figure 9 (from Shaefer and Nikleva, 1973) which shows the mean annual precipitation field for the greater Vancouver area. It is apparent that great variations in annual precipitation occur within a small area around Vancouver and this generally correlates with elevation above sea level. In fact, Shaefer and Nikleva point

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Fig. 7:
Monthly Average Number of Days with Greater than 0·2mm Precipitation

(From the Atmospheric Environment Service)

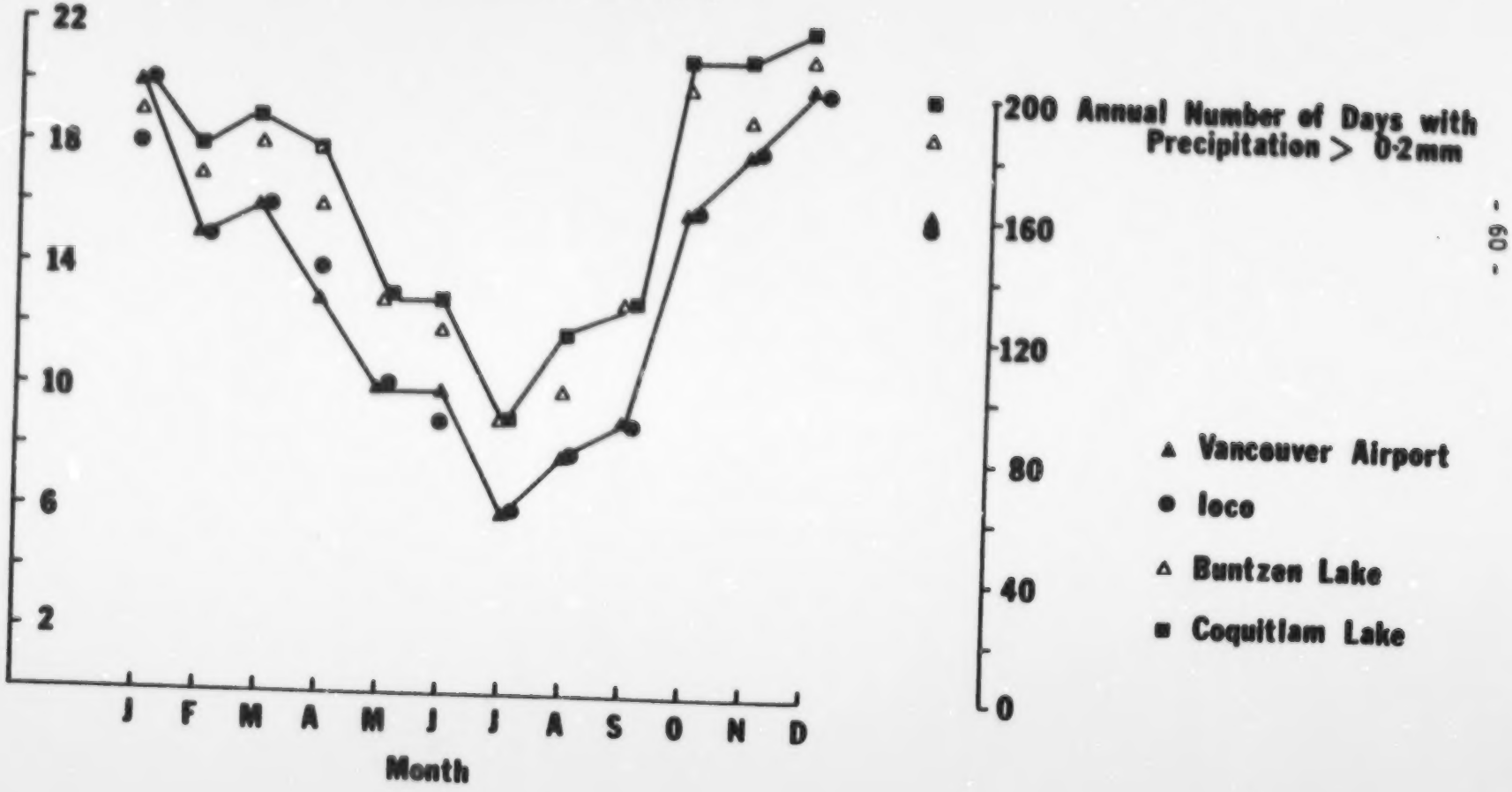

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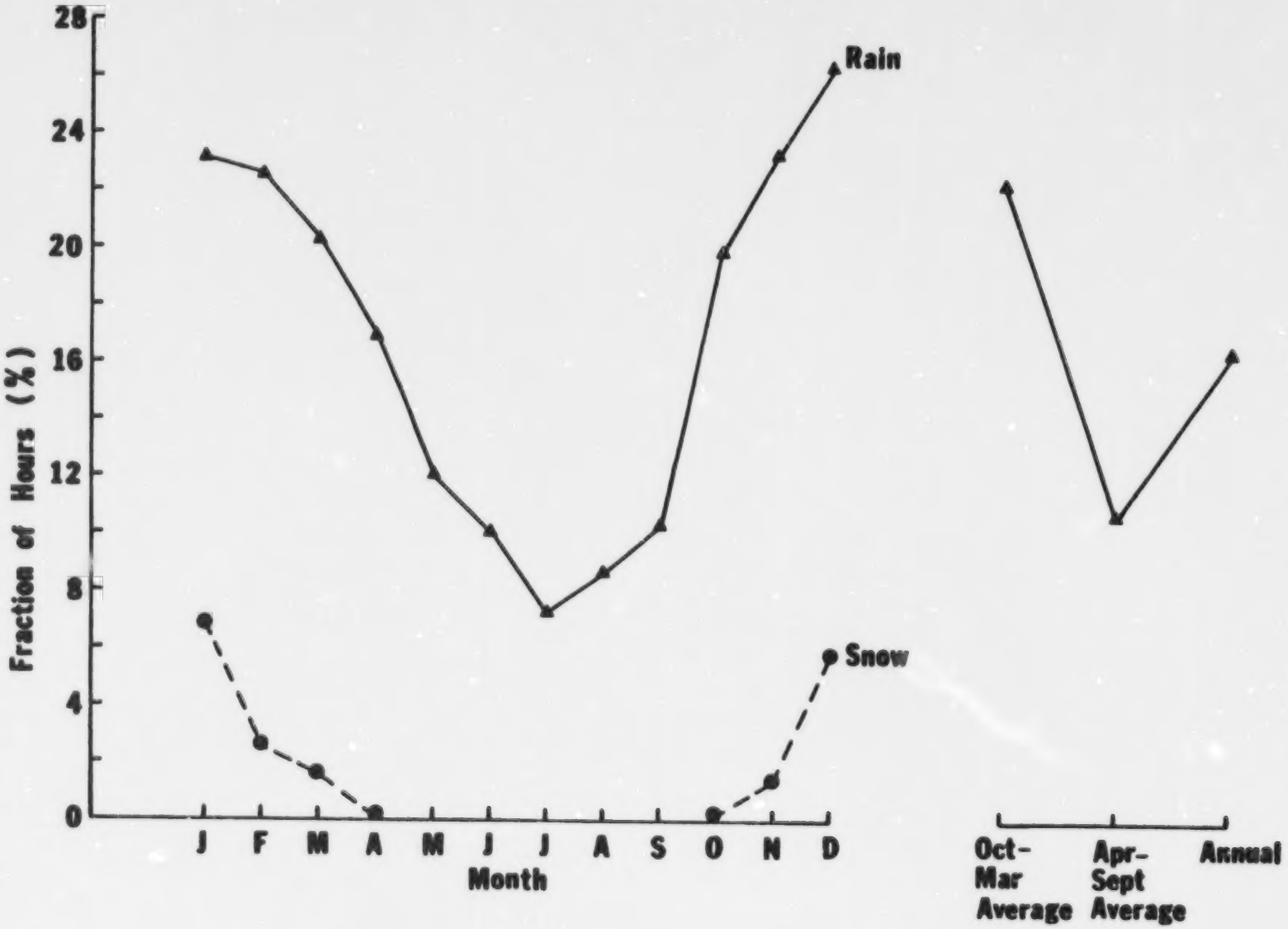
Fig. 8: Fraction of Hours per Month with Recorded Precipitation at Vancouver Airport (20 Year Average)

(From the Atmospheric Environment Service)



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out that there is a southwest-northeast axis throughout the area along which the precipitation increases with distance northward.

It is useful at this point to summarize by indicating that precipitation in the area around BTGP is significantly influenced by orographic effects. Extreme changes in precipitation characteristics occur over very short distances north of the BTGP. This is shown in Figure 6 which illustrates the difference in monthly and seasonal precipitation rates at Vancouver Airport, IOCO, Buntzen Lakes and Coquitlam Lake (the latter three locations being in the vicinity of the BTGP).

Table 6

DETAILS OF PRECIPITATION SAMPLER LOCATIONS

Location	Elevation (m-MSL)	Distance From BTGP (km)	Direction From BTGP	30 Year Average Annual Precipitation (mm)
Vancouver Airport	8.5	25	SW	1068.1
IOCO	27.5	~1	E	2060.1
Buntzen Lake	130.5	7.5	NNE	2738.5
Coquitlam Lake	161	10	NE	3631.2

Table 6 indicates that the elevations of these four sites change quickly and the annual precipitation rate more than triples from Vancouver Airport to Coquitlam Lake.

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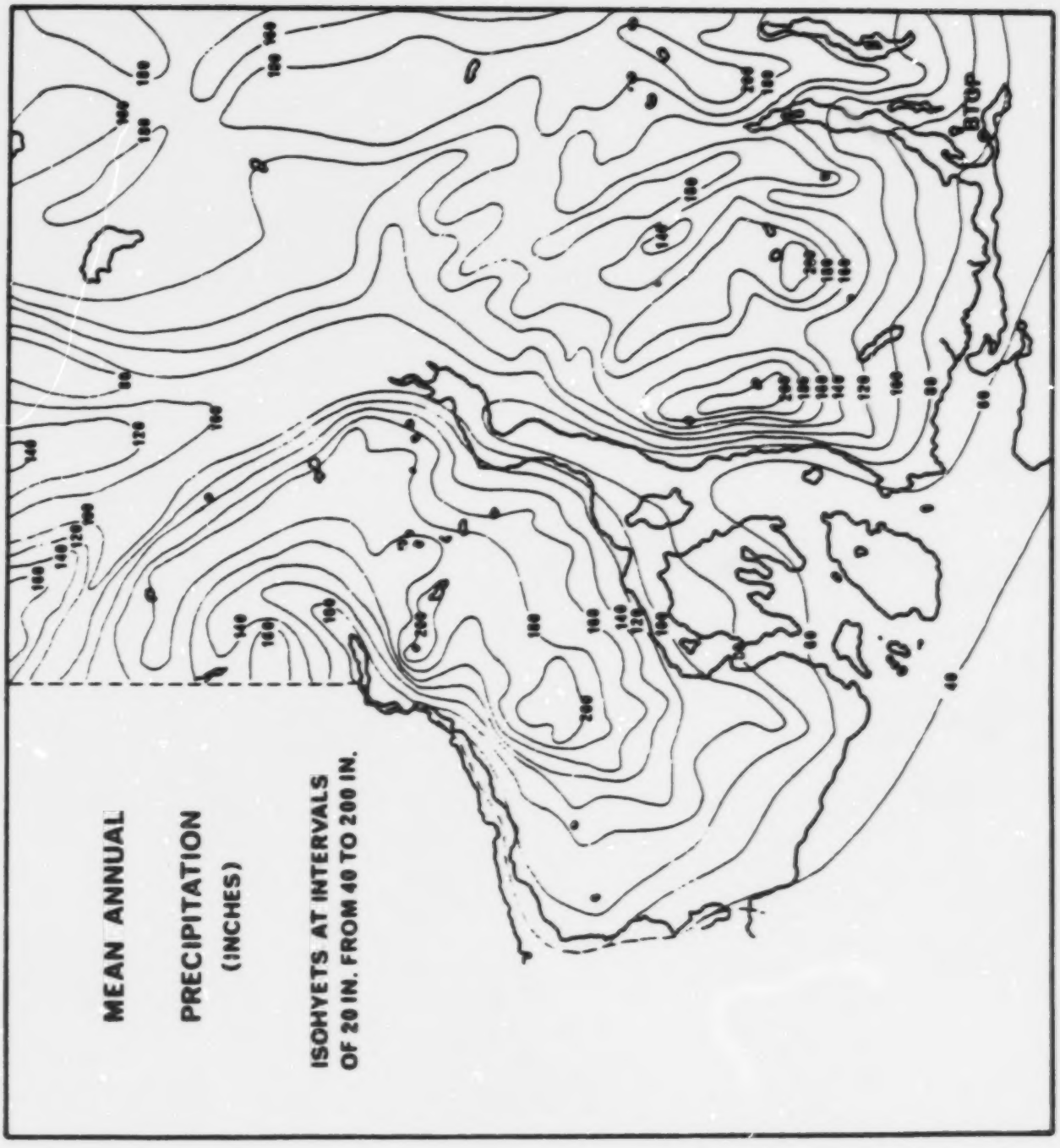

Figure 9: Annual Precipitation Field for the Vancouver Area

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Schaefer and Nikleva (1973) also point out that the differences in total precipitation with elevation are due more to increased intensity of precipitation during events, rather than increased frequency of events. Figure 7 shows that the number of days per year with precipitation over 0.2 mm is relatively constant over the four sites (the largest difference is 25 % between Coquitlam Lake and IOCO). This implies that when precipitation occurs at upper levels, it also usually occurs at lower levels at significantly lower intensities. There is however some enhancement in the frequency of precipitation events at upper levels.

Schaefer and Nikleva also point out that local topography affects the type of precipitation (i.e. rain versus snow) falling in the Vancouver area. They attribute the effect to the depression of temperature with elevation and show that there is an increase in the number of snowfall days per year of roughly 27 days per 300 m of elevation. This manifests itself in the annual snowfall statistics for the area. For example, on average, Burrard Inlet receives roughly 64 cm of snow per year, West Vancouver (elevation 365 m) receives 250 cm and the mountain barriers at the heads of open valleys recieve up to 1800 cm. Hence, when it is raining in the lower portions of Vancouver in the winter, it is generally snowing at the upper levels.

The importance of the precipitation field characteristics for the potential impact of BTGP emissions on wet and dry deposition will be discussed in detail in Chapter 4.

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3.3 Precipitation Chemistry

Very little precipitation chemistry monitoring has been and is being carried out in southwestern British Columbia. The only network of note with easily-accessible data is the Canadian Network for Sampling Precipitation (CANSAP) which is operated by the Atmospheric Environment Service of Environment Canada.

To provide some appreciation for the chemistry of precipitation in the Vancouver area, a brief review of CANSAP data for the Vancouver sampler is presented. It is important to note that the Vancouver CANSAP sampler is located at Vancouver Airport. Because of this, the data cannot be considered representative of regional background precipitation, i.e. given the discussion in Section 3.3.3 on flow directions during most precipitation episodes there will undoubtedly be an urban influence and a local airport-related influence on the precipitation collected. It is suggested therefore, that the CANSAP data be considered as indicative of precipitation chemistry in that specific area but not representative of data for other locations in the Greater Vancouver area, e.g. around BTGP.

A summary of the CANSAP data for acid related species for Vancouver from 1977 through 1979 is presented in Table 7 and a rudimentary analysis of these data is given in Table 8. To provide a basis for comparison, the equivalent volume-weighted concentration data for Port Hardy (on the north tip of Vancouver Island) are presented in Table 8.

By assuming that the Port Hardy data are roughly representative of background precipitation along the west coast of B.C., the Vancouver data can be better interpreted. Note that this assumption is not totally valid but provides a rough basis for comparison. A summary of the data is given below:

1. For the years 1977 through 1979, only 19 of the 36 monthly samples at Vancouver produced usable data.

2. The volume-weighted pH, i.e.

$$-\log_{10} \frac{\sum_i (H^+)_i (V)_i}{\sum_i (V_i)}$$

where V = volume,
i = month,
(H^+) = H^+ concentration in moles/l

for Vancouver for the three years was 4.9. By comparision, the volume-weighted pH for Port Hardy was 5.0, not significantly different from Vancouver's.

3. The volume-weighted concentration of $SO_4^=$ in precipitation was significantly different at Vancouver than at Port Hardy, i.e. 2.76 mg/l compared to 1.32 mg/l. Some of this $SO_4^=$ can

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Table 7

CANSAP DATA FOR ACID RELATED SPECIES FOR VANCOUVER AIRPORT*

Month/ Year	# Days Sampled	Precipitation Depth (cm)	Volume Collected (ml)	pH	$SO_4^=$ (mg/ℓ)	NO_3^- (mg/ℓ)
1/77	-	-	-	-	-	-
2/77	-	-	-	-	-	-
3/77	-	-	-	-	-	-
4/77	-	-	-	-	-	-
5/77	-	-	-	-	-	-
6/77	-	-	-	-	-	-
7/77	-	-	-	-	-	-
8/77	31.0	6.4	1550.0	5.8	1.9	1.02
9/77	32.0	11.4	1660.0	5.0	4.4	3.41
10/77	32.0	6.9	1700.0	4.8	2.4	1.33
11/77	30.0	17.3	3750.0	4.9	2.0	0.80
12/77	32.0	16.5	3570.0	4.9	1.6	0.74
1/78	33.0	12.8	2720.0	4.6	1.6	1.24
2/78	-	-	-	-	-	-
3/78	-	-	-	-	-	-
4/78	-	-	-	-	-	-
5/78	20.0	6.6	870.0	4.9	5.5	2.35
6/78	-	-	-	-	-	-
7/78	-	-	-	-	-	-
8/78	31.0	9.6	2200.0	5.3	2.4	0.98
9/78	31.0	10.6	2080.0	4.7	3.3	1.56
10/78	32.0	4.5	780.0	4.5	5.7	3.40
11/78	31.0	12.1	2640.0	4.6	1.9	0.71
12/78	32.0	9.4	2280.0	4.5	4.7	1.41
1/79	-	-	-	-	-	-
2/79	-	-	-	-	-	-
3/79	34.0	5.7	1750.0	4.6	3.7	1.55
4/79	31.0	5.8	600.0	4.9	6.2	3.15
5/79	-	-	-	-	-	-
6/79	-	-	-	-	-	-
7/79	-	-	-	-	-	-
8/79	34.0	2.1	200.0	6.6	20.6	1.59
9/79	29.0	7.9	1070.0	4.7	3.7	1.77
10/79	32.0	8.0	-	4.7	2.6	1.20
11/79	31.0	8.3	1820.0	4.5	2.9	1.20
12/79	32.0	27.6	6640.0	4.8	1.4	0.44

* Volume-weighted concentrations are corrected for evaporative losses from June to August.

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Table 8

1977 - 79 Volume-Weighted Concentrations for Vancouver

Parameter	Volume-Weighted Concentration* (mg/l except pH) Vancouver	Port Hardy
pH	4.9 (19)**	5.0 (29)
$SO_4^=$	2.76 (19)	1.32 (28)
NO_3^-	1.31 (19)	0.33 (29)

* Volume-weighted concentrations are corrected for evaporative losses from June to August.

** Numbers in parenthesis represent number of monthly samples included in the analysis

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reasonably be attributed to an urban influence on the Vancouver samples but it is difficult to know how much. The same situation appears for the volume-weighted NO_3^- concentrations at the two sites, i.e. 1.31 mg/l at Vancouver versus 0.33 mg/l at Port Hardy.

4. The ratio of $SO_4^=$ volume-weighted concentration to NO_3^- volume-weighted concentration in mass units was 2.11 at Vancouver. On a molar basis, this ratio is 1.36 which indicates that $SO_4^=$ could potentially account for 170 % more of the strong acid contribution to local precipitation than NO_3^- (on an acid equivalent basis).

5. Points 3 and 4 combined indicate that Vancouver has concentrations of $SO_4^=$ and NO_3^- enhanced over background precipitation. However, the enhancement for NO_3^- appears to be greater than for $SO_4^=$.

As a basis for comparison of the acidity of Vancouver's precipitation with areas considered to have significant anthropogenic long range transport effects on precipitation quality, data from the Peterborough, Ontario CANSAP sampler are given below in Table 9. Only pH, $SO_4^=$ and NO_3^- are reported because the former is a measure of the acidity of the precipitation and the latter two are the primary contributors to the acidity.

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Table 9

1977 - 79 Volume Weighted Concentrations for Peterborough, Ontario

Parameter	Volume-Weighted Concentration
pH	4.3
$SO_4^=$	5.43 mg/l
NO_3^-	3.71 mg/l

It can be seen that the volume-weighted concentration of $SO_4^=$ at Peterborough was roughly twice that of Vancouver. Also, the concentration of NO_3^- was roughly 3 times greater at Peterborough than Vancouver and the ratio of $SO_4^=$ to NO_3^- was roughly unity on a molar basis. This indicates that precipitation in the Vancouver area contained less $SO_4^=$ and NO_3^- than that in southern Ontario but still significantly higher levels than in background precipitation as measured at Port Hardy.

In summary, a very limited analysis of CANSAP data for Vancouver suggests the following:

- the levels of $SO_4^=$ and NO_3^- in Vancouver precipitation are, in the long term enhanced over background precipitation with NO_3^- appearing to be enhanced more than $SO_4^=$. This is expected from the emission inventory (Section 3.1).

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- the volume-weighted concentration of $SO_4^=$ is roughly 170 % greater than that of NO_3^- (on an acid equivalent molar basis)

- the concentrations of $SO_4^=$ and NO_3^- are significantly less in Vancouver than in areas of southern Ontario where significant anthropogenic effects on precipitation are noted.

3.4 Air Quality Data

Information on air quality in the Greater Vancouver area is available from the Greater Vancouver Regional District (GVRD) Ambient Air Monitoring Network. The air quality data are important because of the relationship between ground level concentration and dry deposition. These data have been analysed in detail (Concord Scientific Corporation, 1982, 1983) and will not be discussed further here except to note that they will be used in Chapter 4 in assessing the incremental effect of BTGP on dry deposition of acidic species.

3.5 Lake Chemistry

An area of considerable concern is the impact of acidic compound deposition on the water quality of lakes. Acid deposition (both from wet and dry mechanisms) can occur by direct input to lake surfaces but a significant input also arises from the watershed area which drains into the lake. The impact of acid deposition on lake pH

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and alkalinity depends on several factors such as lake volume, flow through rates, surface area of the drainage watershed, and most importantly the type of soil in the area. Basic species in the watershed soils can serve to neutralize the acidity of deposition to a lake. Thus the long term impact of acid deposition to an area is critically dependent on the soil type in the area of interest and its buffering capacity.

The geological conditions in the area to the north east of the Greater Vancouver region are characterized by soils of relatively low buffering capacity and hence lakes in this region can be expected to be fairly sensitive to acid deposition effects.

Considerable effort has been spent studying lakes in Ontario where a mixture of geological conditions produces lakes with very high buffering capacity in the southern regions of the province and low buffering capacity in the north. Because of this many lakes in the northern areas of Ontario are becoming more acidic.

In a recently published survey of more than 1500 lakes in Ontario, all lakes were grouped into one of five categories according to their sensitivity to acid input (Ontario Ministry of the Environment, 1981). The categories used were as follows: (Alkalinity is expressed as eqivalent $CaCO_3$ in ppm).

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"Acidified" Lakes (alkalinity < 0 ppm)

These lakes have zero or negative alkalinity. The pH will usually be less than or equal to 5.0 (some lakes may be this acidic naturally, such as Sphagnum bogs). Fish population will be severely stressed. Many species may be extinct or the lake may be without fish. Lakes that are not naturally high in organic colour, such as beaver ponds, will be exceptionally clear. Clouds of algae may accumulate in the inshore area. Clam, snail, and many amphibian species will be absent.

Extreme Sensitivity (alkalinity 0 - 2.0 ppm)

These lakes have very low alkalinity, but the pH itself will probably not be toxic to most of the biota for most of the year. The pH will almost certainly be depressed during spring snow melt and fish kills and other biological damage may occur at that time.

Moderate Sensitivity (alkalinity 2 - 10 ppm)

These lakes will have acceptable pH values for most of the year. In areas experiencing heavy acid loading the pH may be depressed significantly in certain lakes during snow melt especially if the melt proceeds rapidly. Since the pH will be less frequently depressed than in the extreme sensitivity category, the biota will be less at risk.

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Low Sensitivity (alkalinity 10 - 25 ppm)

Lakes in this category are likely to experience seriously low pH and biological damage only under very extreme snow melt conditions.

Not Sensitive (alkalinity > 25 ppm)

Lakes in this category contain sufficient buffering capacity to neutralize acidic rain for an indefinite period of time. Snow melt will not depress the pH sufficiently to result in biological damage.

The Sudbury district of Ontario is in the Canadian Shield (precambrian rocks) which is characterized by low buffering capactiy. Thus, for this area, of the 208 lakes surveyed 23 % were classed as acidified, 29 % extremely sensitive, 24 % moderately sensitive, 11 % of low sensitivity, and 13 % not sensitive. This contrasts with areas such as Simcoe County in southern Ontario where the geology is characterized by sedimentary deposits and high buffering capacity. For this region 100 % of the lakes tested were considered not sensitive.

In the general region of Vancouver there are available water quality data for a number of lakes (B.C. Hydro Survey data). In particular Seymour Lake, Capilano Lake and Coquitlam Lake have been monitored since 1968. These three lakes are in locations which may experience some impact from the urban emissions from the Greater

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Vancouver area and the BTGP (Capilano Lake, in particular, is in the sector of greatest impact for wet deposition, see Section 3.2). There has been no discernable trend in pH for these lakes which ranged from 6.0 to 6.6 and averaged about 6.3 for all three lakes. There does appear however to have been a slight drop in alkalinity over the period 1968 to 1980. Most of this occurred in the late 1960's and early 1970's.

According to the criteria outlined above, Seymour Lake and Capilano Lake would be described as moderately sensitive to acid input effects and Coquitlam would be classified as extremely sensitive. It should, however, be noted that despite this sensitivity the slight drop in alkalinity observed for these lakes appears to have occrred in the early 1970's and there does not appear to have been a significant change over the past few years. This observation would be consistert with the conclusion that the lake systems in the area are able to tolerate current loading levels of acidic compounds.

Lake water quality data are also available for Stave Lake, Rolly Lake, Whonnock Lake, Alouette Lake, Pitt Lake and Buntzen Lake. Although as extensive a data base is not available here, the current pH levels for these lakes are similar ranging from 6.3 to 7.2. The alkalinity is less than 10 ppm for all of these lakes. Thus they are at least moderately sensitive to effects from acid deposition.

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The following summarizes the important conclusions with regard to lake water quality in the Greater Vancouver area:

1. Most of the lakes in the area are either moderately or very sensitive to effects from acid deposition. This is consistent with the geology of the area.

2. Current loading rates of acid compounds to these lakes do not appear (on the basis of limited data) to be causing a rapid change in water quality as measured by alkalinity levels.

3. pH levels in these lakes average about 6.3 with a range of 6.0 to 7.2. No current trends are apparent in the data.

4. ASSESSMENT OF THE IMPACT OF INCREMENTAL EMISSIONS FROM BTGP ON WET AND DRY DEPOSITION IN THE VANCOUVER AREA

The information presented in Chapters 2 and 3 was designed to provide a basic understanding of the mechanisms of wet and dry deposition as well as an information base for the assessment of the impact of Burrard Thermal Generating Plant emissions on wet and dry deposition in the Greater Vancouver area. In this chapter the assessment is carried out; it is presented under two separate headings -- Qualitative Assessment and Quantitative Assessment. The former is based on present knowledge of the fields of wet and dry deposition as they relate to conditions in the Vancouver area. The latter is based on a simple mathematical model which allows the potential impact of BTGP emissions to be quantified.

By way of introduction to the assessment it is useful to point out that the major determining factor is the strength of the source, which, for the BTGP operating continuously at normal maximum load is approximately 417 kg hr^{-1} or 3.66×10^6 kg yr^{-1} (expressed as NO_2). The estimated total emission of nitrogen oxides also expressed as NO_2 for the Greater Vancouver area was 63.2×10^6 kg in 1976. However, not all of these emissions are capable of impinging on receptors in the vicinity of the BTGP. The appropriate fraction is not certain, since a geographical breakdown of emissions is not available, but if we assume

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that roughly half of the Greater Vancouver emissions can reach the Burrard Inlet area it can be calculated that the BTGP contribution would be about 12 %.

The concentration of a particular pollutant reaching the receptor is also dependent on the rate of the chemical reactions taking place in the atmosphere, and on the amount of dispersion which occurs. In turn, the reaction rates are affected by background pollutant concentrations, since, as was discussed in Section 2.1, aged, polluted air will normally contain elevated concentrations of ozone and free radicals (such as HO_2 and RO_2) which enhance the rate of oxidation of the nitrogen oxides. Typically, NO is oxidized at 20 - 60 % per hour, while the rate for NO_2 is about 15 % per hour. These rates will apply as long as the oxidizing species are present. Peroxy and other free radicals normally disappear at sundown, but ozone may persist after dark.

4.1 Qualitative Assessment

4.1.1 Wet Deposition

This part of the assessment is based on a knowledge of the state-of-the-art in the field of wet deposition as it applies to the case under study. It is approached by first presenting results of

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experimental studies of precipitation scavenging of locally-emitted pollutants; then by discussing BTGP's situation in light of the background information presented in Chapters 2 and 3.

4.1.1.1 Experimental Research on Precipitation Scavenging of Locally-Emitted Pollutants

To date, very few experimental studies on precipitation scavenging of locally-emitted pollutants have been carried out. Of those that have, most have focussed solely on the scavenging of sulfur oxides and trace metals, e.g. Granat et al (1975), Chan et al (1982), Millan et al (1982), Enger et al (1979) and Wiebe et al (1974). Very little attention has been devoted to the scavenging of nitrogen oxides (which is of direct application to this assessment since NO and NO_2 represent the only significant emissions from the BTGP). Only two studies on the latter topic exist and the results of both are extremely important to this assessment. They concern emissions from:

a) the Centralia Steam Electric Plant of Pacific Power and Light Company in Washington State studied by Dana, Drewes, Glover and Hales (Dana et al, 1976),

b) the city of St. Louis studied by Hales and Dana (Hales and Dana, 1979).

Each is discussed briefly in this section.

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a) Centralia

A series of field experiments to measure the effect of emissions from the Centralia Steam Electric Plant on local precipitation chemistry was carried out in December 1974 by Battelle Pacific Northwest Laboratories. The Centralia plant is located in SW Washington state and is a dual-unit, coal-fired station with a maximum output of 1300 MW. Precipitation samplers were placed in both cross-wind and down-wind orientations at distances between 0.4 and 11 km from the stacks. Three precipitation events were successfully sampled and the precipitation collected was analyzed for SO_2 (dissolved), $SO_4^=$, NO_2^-, NO_3^-, total inorganic fixed nitrogen, NH_4^+, and PO_4^{3-}.

The results of the programme can be summarized briefly as follows: (1) sulphate concentrations in rainwater due to the plume's presence were not significant at downwind distances less than approximately 10 km; (2) SO_2 and hydrogen ion showed significant plume-related deposition patterns at all distances from the stack; (3) nitrite and nitrate were at or near background levels and showed negligible plume-related scavenging.

The implications of these results are quite important. First, they clearly suggest that the scavenging process is very inefficient for NO and NO_2 under relatively clean background conditions. This result is consistent with recent work carried out by Levine et al (1980) who show that Henry's law coefficients for NO and NO_2 are very low.

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The study results also imply that the oxidation of NO_x emissions to HNO_3 (which is very efficiently scavenged) was slow within the first 11 km of the stack. This too is consistent with the clean background air mass and the conversion mechanisms for NO to NO_2 to HNO_3 presented in Chapter 3.

b) St. Louis

During the summer periods of 1972 and 1973 a precipitation sampling program was carried out in the area of St. Louis, Mo. The sampling was designed to assess the effectiveness of scavenging urban pollutants by convective storms.

Samplers were located upwind and downwind of the city in a distribution designed to measure pre-city and post-city precipitation quality. The precipitation samples were analysed for $SO_4^=$, SO_2, NO_3^-, NH_4^+ and H^+.

The results of the study were as follows:

i) Based on a regional mass balance approach, it was estimated that 36 % of the locally-emitted sulfur and 13 % of the locally-emitted NO_x was scavenged within about 48 km of the city.

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ii) The amount of non-locally emitted $SO_4^=$ and NO_3^- deposited during the same storm was very high and was mostly attributable to direct scavenging of regional $SO_4^=$ and NO_3^- aerosols. In other words, the background air mass approaching St. Louis was highly polluted and the pollutants had transformed in the aged air mass to more easily scavenged compounds.

The implications of these results are quite important, especially in light of the Centralia results (for relatively clean regional air). They suggest that the high deposition of locally-emitted sulphur oxides was due to the relatively fast conversion of SO_2 to H_2SO_4 or metallic sulphates with subsequent scavenging of these compounds. This is supported by the facts that levels of SO_2 in the precipitation were low and that oxidizing agents such as ozone and hydrogen peroxide were probably generated by photochemical activity before the storm. It is known that aqueous-phase oxidation can result in rapid conversion of SO_2 to $SO_4^=$ (see Section 2.2.2).

The most important implication of the study, however, relates to the measured levels of NO_3^- in the precipitation. While the amount of scavenged NO_3^- attributed to the urban emission was relatively small (13%), it was still large when one considers that both experiment (Dana et al, 1976) and theory (Levine and Schwartz, 1980) indicate that negligible amounts of NO and NO_2 are scavenged in the vicinity of sources under clean background conditions. It was therefore felt that the significant scavenging of NO_3^- from the urban source in this study

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was due to chemical conversion of NO_x to more easily soluble products such as HNO_3 and PAN. These conversion processes were suggested to have photochemical origins but this was not clear from the study. The result, however, is quite apparent; that is, with high levels of pollutants in the background and urban air masses, precipitation scavenging of sulfur and nitrogen oxides can be higher than for the same pollutants emitted into clean air masses.

One final point to indicate in this latter study is that the concentration of H^+ ion in the precipitation did not correlate well with NO_3^-, $SO_4^=$ or NH_4^+. This appears to reflect the abundance and mix of other ionic species in the St. Louis rainwater and suggests that the enhanced acidity of rain in the urban air was due to many sources - not just the major SO_2 and NO_x sources.

4.1.1.2 Assessment

The impact that the BTGP emissions will have on wet deposition in a particular receptor area will depend on the following factors:

a) The chemical form and the source strength of the emission.

b) The conversion of primary emissions between exit and arrival at the receptor and the corresponding formation of secondary pollutants.

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c) The concentration and height of the emissions at and before the receptor (with respect to plume rise and dispersion).

d) The frequency with which the emissions travel to the receptor.

e) The background levels of pollutants (from the point of view of causing faster conversion of the primary emissions as well as direct contribution to background levels of wet deposition).

f) The fraction of emissions compared to the total emission inventory for the source region.

g) The frequency of precipitation episodes.

h) The characteristics of the precipitation.

i) The efficiency of precipitation scavenging of the emitted and background pollutants.

j) Other secondary factors such as the contribution of scavenged emissions to pH compared to other scavenged materials.

The only emissons from the BTGP which can potentially impact on the local wet deposition field are of NO and NO_2.

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Clearly, any impact would be a contribution to the concentration of nitrogen compounds and free hydrogen ion. However, as mentioned in Section 2.5.1, the precipitation scavenging efficiency for NO is extremely low (because of its low solubility in water) and the efficiency for NO_2 is only slightly higher. In fact, the plume washout experiments at the Centralia Power Plant (Dana et al, 1976) showed clearly that negligible NO_2 was scavenged from a coal-fired power plant plume in relatively clean background air.

Given that theory and experiment both provide strong evidence that the primary BTGP emissions will not affect the deposition of nitrogen compounds or the acidity of local precipitation, attention must be focussed on the other possible route for the emissions to affect precipitation, i.e. through the formation and scavenging of HNO_3 and/or PAN. Both HNO_3 and PAN are known to be highly soluble in water and their scavenging efficiencies are quite high.

The discussion of NO_x chemistry in Chapter 2, showed that the conversions of NO to NO_2 and NO_2 to HNO_3 and PAN are highly dependent on the presence of ozone, hydrocarbons and/or photochemical activity in the air mass into which the emissions are released. A reasonable rate of conversion of NO_2 to HNO_3 in an urban plume is roughly 15 % per hour (Spicer, 1979a, 1979b). Table 3 shows the expected rate of conversion of emitted NO to HNO_3 and PAN for both clean and urban air. It shows that the fraction of NO emissions converted to HNO_3 and PAN after an

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hour's travel time is roughly 3 % under clean background conditions and 6 % under polluted conditions. Even after 5 hours, only 30 % is converted in clean air and 50 % in urban air.

These numbers allow us to explore the influence of the BTGP emissions on precipitation under several meteorological scenarios which apply to the Vancouver are:

a) Frontal Precipitation

The Pacific frontal systems which account for most of Vancouver's precipitation are characterized by moderate to strong winds from the southeast-east-northeast sector and precipitation rates which increase signficantly to the north of the city. In terms of the impact of BTGP emissions on local wet deposition, it is expected that the BTGP emissions would most frequently be channelled to the west along Burrard Inlet with less frequent travel to southeast and northwest. The background air into which the BTGP plume is emitted would be relatively clean - cleanest with northeast flow and not quite as clean with southeast flow. In the latter case, emissions from the suburban areas east and southeast of BTGP would affect the air mass. This relatively clean background air is important for two reasons. First, it ensures a low conversion rate of BTGP's primary NO and NO_2 emissions to HNO_3 and/or PAN. This in turn, limits the amount of nitric acid available

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for precipitation scavenging in the vicinity of BTGP (NO and NO_2 will not be efficiently scavenged themselves). Second, the acidity in the precipitation will be low compared to that in a polluted air mass.

After emission into the clean background air, the BTGP plume will travel, at reasonably high wind speeds, towards the City of Vancouver, where it will become imbedded within the urban NO_x emissions (BTGP emissions account for 6 % of the GVRD emissions). As the levels of urban emissions increase, the conversion of BTGP primary emissions will speed up and precipitation scavenging will become more effective. However, the amount of BTGP NO_x scavenged relative to the urban-generated emissions will remain low because of the higher emission rate of the latter. Given the relatively high wind speeds during these events, i.e. 14 to 20 km h^{-1} (see Appendix 1), and the large induction period for BTGP emissions (see Section 2.4), the BTGP NO_x emissions will probably not begin to be scavenged until they are well over the City of Vancouver or Vancouver Harbour. At this time, the NO_x conversion rate will begin to increase due to the urban emissions and the scavenging of BTGP and urban-related NO_x will increase. It is likely, however, that most of the BTGP and urban emissions will be transported over the Strait of Georgia or Howe Sound within one to two hours of emission - thereby limiting the amount of NO_x available for precipitation scavenging to that which could be converted to HNO_3 in that time period. Based on the St. Louis experiment, which was more conducive to NO_x scavenging than Vancouver because of the polluted

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background air and very high urban emissions, it is expected that the scavenging of BTGP NO_x in this time period would be low - less than 13 % and would be a relatively small fraction of the urban emissions scavenged at the same time.

In summary, the characteristics of frontal storms plus wind speed and wind direction statistics for precipitation periods suggest that the main impact of BTGP emissions in the local area would begin over the city and harbour of Vancouver after which the emissions would typically be transported over the Strait of Georgia or to the areas northwest of the city. The nature of the scavenging mechanisms for NO, NO_2 (inefficient) and NO_3^- (more efficient) plus the slow rate of conversion of primary NO_x emissions in the relatively clean background air suggest that BTGP emissions will not be scavenged until they are over the city. Even then, the fraction of BTGP NO_x expected to be scavenged (based on the St. Louis experiment) should be significantly less than 13 % and this will be a small fraction of the acid-related materials scavenged from other urban emissions.

In terms of the potential incremental impact of BTGP emissions on local lakes, it is those lakes to northwest of BTGP that will experience the greatest, long-term potential impact - particularly Capilano Lake. These lakes are in the mountains where precipitation rates are significantly higher than in the city. This has important ramifications

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on precipitation scavenging, as extreme variations in precipitation intensity in the area have a direct effect on both the concentration of pollutants in the precipitation and the total amount of material wet deposited. Based on both microphysical storm dynamics and mass deposition arguments, there is clear evidence (Hales and Dana, 1979; Barrie, 1981) that higher precipitation intensities result in lower pollutant concentrations. However, even though the concentrations are lower the total wet deposition of the pollutants is higher because of the greater precipitation amount. Hence, one would expect that as air moves into the mountainous high-precipitation-intensity area north of Vancouver, the concentration of pollutants in the precipitation would decrease, but the net pollutant deposition to the area would increase. This implies that the high precipitation areas northwest of BTGP are better protected from high concentration episodes than other areas, but would tend to have slightly higher long-term deposition rates. Acid shock due to spring melt would therefore be more important in these areas.

It was shown in Chapter 3 that, in the winter, the precipitation generally changes from rain to snow at higher elevations. There is presently no information available on the effect of phase change (i.e. rain or snow) on the scavenging efficiency of HNO_3; however, the potential for an effect does exist.

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b) Pre-Frontal Precipitation

The Pacific frontal systems referred to earlier are generally preceeded by strong easterly or southeasterly flow. This flow is important because it tends to transport the high levels of pollutants, which can accumulate in the area, out of the region prior to the onset of precipitation (Emslie, 1981). This means that high air pollution episodes due to stagnation or oscillating land breeze - sea breeze situations do not have much effect on local precipitation quality.

c) Convective Precipitation

Schaefer (1981) indicated that convective precipitation is relatively infrequent in the Vancouver area. Since no climatological information is available on its frequency, it is not addressed in detail here.

To this point the assessment has been almost totally qualitative. The conclusion is that primary emissions from the BTGP will not be scavenged efficiently from the plume. In order to be scavenged, the NO and NO_2 emissions must convert to HNO_3 or PAN. Experimental and theoretical work has shown that in an urban situation such as Vancouver's, the conversion rate of NO_x to these secondary products can be as high as 15 % per hour and the scavenging of of NO_3^- within roughly 50 km could vary from almost zero to as much as 13 % of the local

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emission inventory. It was pointed out that the situation where the higher scavenging rate was found was expected to be much more conducive to fast conversion of NO_x than in Vancouver. It was felt that BTGP emissions would predominately travel towards the city of Vancouver and the amount scavenged would be relatively small.

The assessment so far has focussed solely on the fraction of the BTGP's emissions that are expected to impact locally. It has not addressed the problem of what this fraction of total emissions could add to the background wet deposition field. The latter is addressed in a more quantitative manner in Section 4.2.

4.1.2 Dry Deposition

The primary pollutant emitted by BTGP, NO, can be neglected in assessing incremental effects of the plant on dry deposition. The reason is that NO, as far as has been demonstrated to date, has a low deposition velocity. Fluxes of NO to the surface are consequently somewhat smaller than those of other species.

The impact of the BTGP plume depends critically on meteorological conditions, so that, in the long term, the areas of impact are widely distributed. The incremental effect of the BTGP on air quality is then obtained as the incremental addition to the emissions of the area. For NO_2 this increment could be approximately 12 % for operation of the plant at normal maximum load (see Section 3.4).

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The incremental effect on dry deposition of NO_2 is the same as the effect on atmospheric concentrations, because the deposition process is roughly linear with pollutant concentrations. To put the dry deposition into perspective, it should be noted that the total amount of NO_2 deposited by this mechanism in the Greater Vancouver area is approximately equal to the total amount of dry deposited SO_2, even though the ambient concentrations of NO_2 are generally approximately double those of SO_2. The compensating factor is the smaller deposition velocity of NO_2, as discussed in Section 2.3.2.

To restate this in another way, the total impact of NO_2 brought to the surface of the earth by dry depositon is limited by the relative inefficiency of the process. Of the amount which is deposited, the contribution from the BTGP is, in the long term, small.

The conversion rate of nitrogen oxides to nitric acid was estimated in Section 2.4, and it was found that in the worst expected case (i.e. most rapid transformation) a reaction time of 4 hours is required for conversion of 40 % of the initial nitric oxide to nitric acid. Under conditions of good dispersion, and a typical wind speed of 8 m sec^{-1}, the corresponding distance travelled is 115 km. At such a distance the contributions from the BTGP will merely amount to a fraction (about 6 %) of the total concentration of nitric acid derived from the Greater Vancouver area. The actual concentration of nitric acid will be very small, as a consequence of the dilution experienced

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over such a distance. (For long range transport it is appropriate to consider the ratio of the BTGP emissions to those of the whole Greater Vancouver area).

If dispersion conditions are poor, any particular parcel of pollutants will not travel as far in the reaction time of 4 hours as indicated above; greater acid deposition may therefore be possible in the Greater Vancouver area. However, it will still be true that the contribution from the BTGP will amount to a small fraction of the total concentration and deposition of nitric acid.

Again, this assessment has been totally qualitative. It indicates that the fraction of BTGP emissions dry deposited in the local area will be small and that, in the long-term, the incremental effect over background will be small.

4.2 Quantitative Assessment

While the qualitative assessment presented above provides a relatively clear picture of the potential impact of BTGP's emissions on the local wet and dry deposition fields, it does not provide a good feel for the actual numbers involved. In this section, therefore, the magnitude of the impact that could be due to BTGP is estimated. The estimate is based on the development of a simple mathematical model which was designed to:

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1. Illustrate how the atmospheric chemistry and deposition mechanisms interact and what factors affect them.

2. Show the relative magnitude of the two mechanisms when applied to a single emission source.

3. Provide a quantitative estimate of the potential impact of BTGP emissions on the wet and dry deposition fields in the Vancouver area.

The development of the model is given in Appendix 2. It is a simple box model based on first-order deposition mechanisms and an approximate first-order NO to HNO_3 conversion rate.

It is important to note that the model is best considered to represent long-term average conditions based on the premise that fluctuations of meteorology and source conditions are smoothed out over long periods of time (of the order of a year). This premise allows simplification of the physics involved in the model but means that it applies to average conditions only. The model is useful in providing numerical results, however these should be interpreted with care and must not be considered as highly precise because of the averaging and the uncertainties in the model parameterizations.

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The equations which express the fraction of the BTGP emissions dry and wet deposited within a radius, r, of the stacks are obtained from Equations (11) and (12) of Appendix 2 after allowing for an initial distance travelled during which no HNO_3 is produced (corresponding to the induction period).

It can be seen that the relationship between conversion rate, scavenging rate and dry deposition rate is quite complicated - even in this relatively simple formulation. Careful scrutiny of these equations shows that the model results are highly sensitive to all of the physical parameters - especially the chemical and physical rate constants.

The model was applied to the BTGP situation by substituting reasonable values for the conversion rate constant, k, the induction period, t_i, the precipitation scavenging coefficient, λ_w, and the dry deposition velocity, v_d. To assess the results of the model and to provide a feel for its sensitivity to the chosen values, three situations were investigated:

Case 1: Conditions highly conducive to wet and dry deposition

Case 2: Conditions moderately conducive to wet and dry deposition

Case 3: Conditions poorly conducive to wet and dry deposition.

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The values of the parameters used are summarized in Table 10. The quantity F represents the fraction of emissions available for scavenging.

Several points are of note:

a) Precipitation was assumed to occur 20 % of the time on an annual basis. This is somewhat higher than the value shown in Figure 8 of Chapter 3 for Vancouver Airport. The corresponding value of F is 0.2.

b) Dry deposition is assumed to occur at all times. F is therefore taken to be 1.0.

c) Values of the NO to HNO_3 conversion rate constant, k, and the induction period, t_i, were based on a best fit curve as discussed in Chapter 2.

d) The value of the scavenging coefficient, λ_{w2}, used in the high deposition case (Case 1) was based on highly-efficient in-cloud scavenging of HNO_3. It is roughly 15 times greater than the value of λ_{w2} used in the low deposition case, which was based on below-cloud scavenging.

e) The scavenging coefficients used in the model do not truly reflect the characteristics of the precipitation field in the

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Table 10

Values used in the Model Calculations

CASE	PARAMETER	VALUES WET	VALUES DRY
1. High deposition	Zi(m)	250	250
	$\lambda_{w2}(s^{-1})$*	10^{-3}	0
	$k(s^{-1})$**	4.4×10^{-5}	4.4×10^{-5}
	$V_{d2}(s^{-1})$+	0	0.01
	u (ms^{-1})	5	5
	F++	0.2	1.0
	t_1 (min)	40	40
2. Moderate deposition	Zi(m)	500	500
	$\lambda_{w2}(s^{-1})$*	10^{-4}	0
	$k(s^{-1})$**	3.3×10^{-5}	3.3×10^{-5}
	$V_{d2}(s-1)$+	0	0.01
	u (ms^{-1})	6.5	6.5
	F++	0.2	1.0
	t_1 (min)	50	50
3. Low deposition	Zi (m)	1000	1000
	$\lambda_{w2}(s^{-1})$*	0.65×10^{-4}	0
	$k(s^{-1})$**	2.2×10^{-5}	2.2×10^{-5}
	$V_{d2}(s^{-1})$+	0	0.01
	u (ms^{-1})	8	8
	F++	0.2	1.0
	t_1 (min)	60	60

* From Schwartz et al (1981)

** From Section 3

\+ Assumed values - not available in the literature

++ From AES Climatological Records

Vancouver area. Because the scavenging coefficient increases with rainfall intensity, it should, in fact, reflect the large increases in precipitation intensity with distance north of BTGP. However, this applies only to the low and moderate deposition cases because the value used in the high deposition case is based on the maximum capability of a cloud to deliver water (Levine et al, 1981).

f) No information is currently available on the deposition velocity for HNO_3. It is expected that newly formed HNO_3, being in low concentration, will remain in the gas phase, and, since it is highly soluble in water, it has been assumed that the value of v_d is the same as that for SO_2, i.e. 1 cm s^{-1}. In view of the uncertainties involved in this whole approximate modelling effort there seems to be little point in arbitrarily assigning a possible range of values to v_d, and it has therefore been held constant in all three cases.

g) Since no HNO_3 is formed during the induction period, there is no deposition within a radius of ut_i.

The results of the model calculations are given in Table 11 and Figure 10. The following should be noted:

a) On an annual basis, the fraction of BTGP-emitted NO wet and dry deposited as HNO_3 is roughly equivalent within a radius of 50 km.

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b) Within a radius of 25 km the fraction of NO emissions wet deposited as HNO_3 ranges from 0 % to 0 % to 1.4 % for the low, moderate, and high deposition cases.

c) The total fraction of annual BTGP emissions wet deposited in a year does not exceed 20 % because it is assumed that precipitation occurs only 20 % of the year. It takes over 200 km of travel to scavenge these emissions.

d) Experimental evidence form the Centralia and St. Louis studies support the estimates provided by the model.

e) The fraction of NO dry deposited on an annual basis, within a radius of 25 km varies from essentially zero in the low deposition case to 0.6 % in the high case. Within 200 km, only 4 % at minimum and 47 % at maximum has been dry deposited. This suggests that 100 % depletion of the emitted NO would occur only after extremely long travel times. It should be kept in mind that these figures are based on an assumed deposition velocity of 0.01 m/s for HNO_3 and a negligible deposition velocity for NO and NO_2. This again is conservative because it emphasizes HNO_3 formation. Also, the model formulation assumed that the plume was always in contact with the ground.

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Table 11

Fraction of BTGP NO Emissions Deposited as HNO_3
Within a Given Radius of the BTGP

Radius (km)	FRACTION (%)					
	CASE 1 Maximum Deposition		CASE 2 Moderate Deposition		CASE 3 Low Deposition	
	Wet	Dry	Wet	Dry	Wet	Dry
10	0.0	0.0	0.0	0.0	0.0	0.0
15	0.1	0.3	0.0	0.0	0.0	0.0
25	1.4	0.6	0.0	0.0	0.0	0.0
50	5.0	4.1	0.6	0.7	0.1	0.1
100	10.4	16.9	3.0	4.1	0.9	0.8
200	16.0	46.7	8.7	15.9	4.0	4.0

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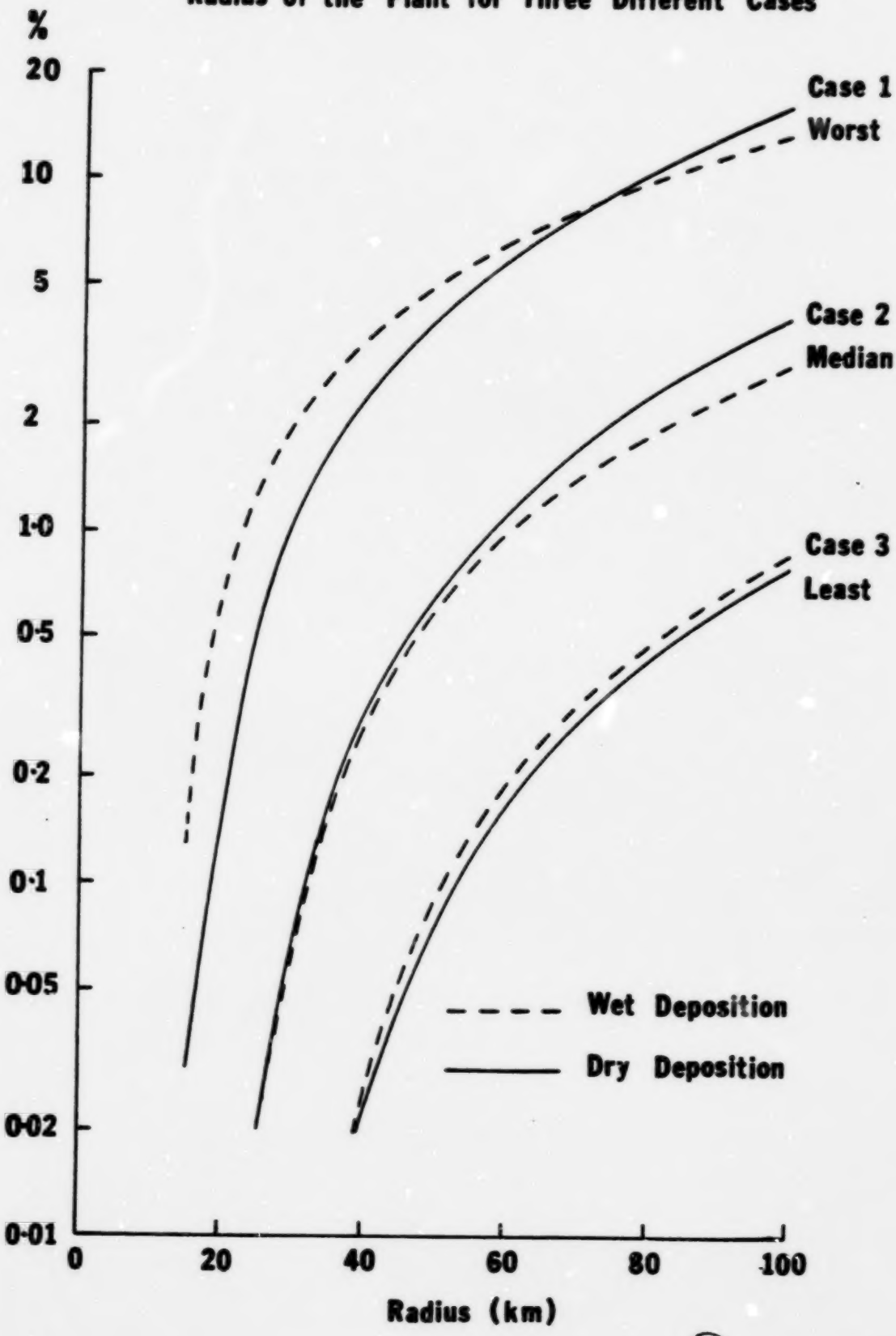
Fig 10: Percentage of Emitted NO Deposited Within a Given Radius of the Plant for Three Different Cases



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In summary, the simple model formulated here appears to compare reasonably well with experimental data in the low and moderate deposition cases but overpredicts in the high deposition case.

The results suggest that over a long averaging period, the fraction of the BTGP NO emissions wet and dry deposited as acid within 50 km of the plant would be quite low, i.e. certainly less than 5 % and probably less than 2 %.

These model results support the qualitative assessment presented in Section 4.1, i.e. a low fraction of BTGP's emissions will be wet and dry deposited. However, to provide some perspective on the incremental impact of BTGP's emissions on the existing wet and dry deposition fields in the Vancouver area, the model results are used in another simple calculation which compares them to background wet and dry deposition rates. This is done for the dry and wet cases in turn.

4.2.1 The Incremental Effect of the BTGP's Emissions on Existing Dry Deposition Rates

If the BTGP operates continuously at normal maximum load, NO emissions will be approximately 3.66×10^6 kg per year. Within a radius of 25 km, which encompasses Buntzen and Coquitlam Lakes to the north of BTGP as well as the major urban areas of Greater Vancouver, Table 11

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shows that in the worst case 0.6 % of the NO emitted will be dry deposited as HNO_3. This corresponds to an average deposition rate of HNO_3 of 0.014 g m^{-2} yr^{-1}.

However, the winds at the BTGP have prevailing easterly and westerly components, so that the impact of the plant will not be uniform within the given 25 km radius. If it is assumed that all emissions travel equally within a 90° sector to the east, and 90° sector to the west, an upper limit to the deposition rate of HNO_3 of 0.028 g m^{-2} yr^{-1} is obtained (within the sectors affected).

It is difficult to put these estimates into perspective, since background HNO_3 concentrations are not known for Greater Vancouver. However, if the mean residence time within the 25 km radius is 3 hours (corresponding to 2.5 m sec^{-1} winds) and the rate constant for oxidation of NO_2 to HNO_3 is 0.15 hr^{-1} (see Section 2:4), approximately 40 % of the ambient NO_2 will be converted to HNO_3. A yearly average NO_2 concentration for 1979 is available for only four of the GVRD stations; the mean of these values is 0.02 ppm. For comparative purposes we therefore suggest an annual average background concentration of HNO_3 of 0.008 ppm. Combining this with the assumed deposition velocity of 1 cm s^{-1}, a background deposition rate of 6.6 g m^{-2} yr^{-1} results. The implication of this is that the incremental effect of BTGP on HNO_3 dry deposition in the Greater Vancouver area will be no more than 0.028/6.6, which is 0.4 %. This value of 0.4 % applies only to the two 90° sectors

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in which the plume travels. Similar calculations were carried out for the other cases considered in the model, and for different distances, assuming a constant background deposition rate. The results are summarized in Table 12.

It should be stressed that these calculations appropriately represent long-term average (e.g. one year) incremental dry deposition amounts. Care should be taken when interpreting the results because of the assumptions used in the model and in the background HNO_3 deposition calculations.

The foregoing calculations are presented somewhat differently in Figure 11, which illustrates the incremental effect of BTGP on dry deposition of nitric acid at a particular distance, as a function of distance. Figure 11 also presents curves for wet deposition of nitrate and hydrogen ion, which are discussed in Section 4.2.2.

The effect of the induction period for nitric acid production is evident in Figure 11, where, even for the high deposition case, no dry deposition of nitric acid occurs for the first 15 km of plume travel. Maximum incremental effect takes place at between 25 and 50 km for BTGP (depending on the conditions) and thereafter remains constant, or decreases slowly. This occurs because the deposition of increasing amounts of HNO_3 is offset by the increasing area to which it is deposited.

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Table 12

Per Cent Increase in the Dry Deposition of HNO_3
Within a Given Radius Due to Operation of the BTGP

Case	Radius (km) 10	25	50
Case 1: High	0	0.4	0.8
Case 2: Moderate	0	0	0.1
Case 3: Low	0	0	0

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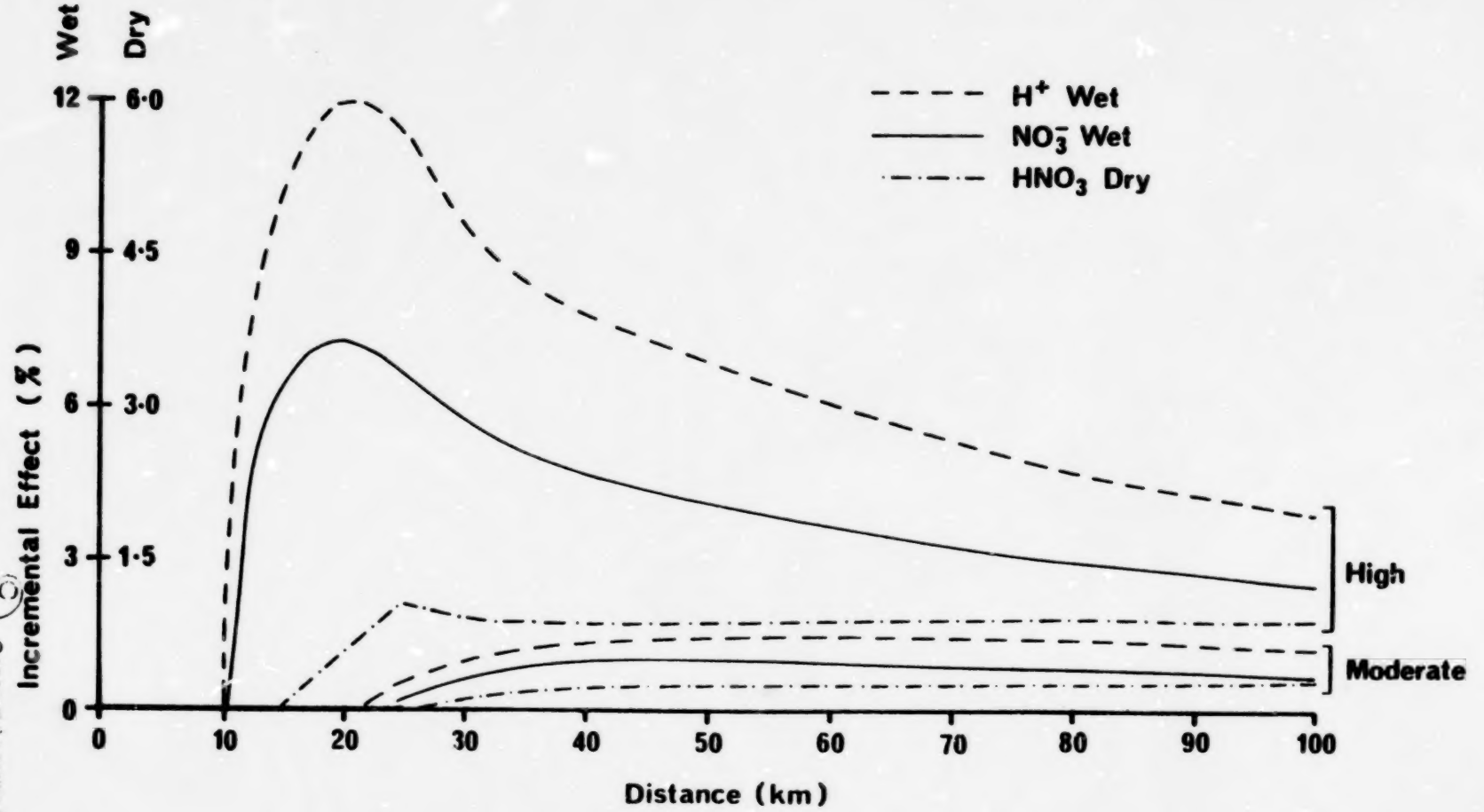
FIGURE 11: INCREMENTAL EFFECT OF BTGP ON DEPOSITION AS A FUNCTION OF DISTANCE FROM THE PLANT



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Note that the incremental effect at a particular distance is larger than the integrated incremental effect calculated within a certain radius of the plant. This occurs because there is essentially no deposition of nitric acid within the induction period. Even so, the maximum incremental effect at a given point, calculated for conditions conducive to high deposition, is only 1.1 %.

The contributions calculated above are somewhat smaller than those derived from the ratio of BTGP emissions to the Vancouver emission inventory. The difference reflects in part the assumptions inherent in the model, but may also be attributed to the fact that essentially no HNO_3 is formed in the BTGP plume for a substantial portion of its travel, corresponding to the induction period. Urban emissions on the other hand, will tend to react faster, as discussed in Chapter 2, and will thus have a shorter induction period.

4.2.2 The Incremental Effect of the BTGP's Emissions On Existing Wet Deposition Rates

In determining the incremental effect of BTGP emissions on local wet deposition rates, it is necessary to determine first the magnitude of existing deposition rates. These existing rates are referred to as background rates in the balance of the text and they consist of both true background and urban-influenced deposition rates. Unfortunately, the only wet deposition data that exist for Vancouver are

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from the CANSAP sampler at Vancouver Airport (see Section 3.4). Under most precipitation conditions, these data will be affected by urban emissions (because the sampler is downwind of many of the urban emission sources). It is therefore expected that pollutant concentrations and deposition at Vancouver Airport will be greater than those typically occurring in the area of BTGP. (The St. Louis study, referred to earlier, supports this contention).

The Vancouver Airport precipitation chemistry data were used to prepare the data in Table 13, which summarizes the expected incremental effect of BTGP on wet deposition, with respect to NO_3^-, H^+ and pH.

The background deposition rate was obtained by combining the three year volume-weighted mean concentration at Vancouver Airport (1.31 mg l^{-1} for nitrate, and 1.26×10^{-5} mol l^{-1} for H^+ - corresponding to a pH of 4.9) with an assumed precipitation depth of 2 m for the impacted areas. (i.e. an approximate average depth across the 90° sector west of BTGP-see Figure 9). The incremental wet deposition from the BTGP was assumed to occur within a 90° sector to the southwest-west-northwest out to the appropriate radius. For ease of comparison, the per cent incremental impact on hydrogen ion concentration was also expressed as a number of units depression in pH.

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Table 13

Annual Average

Incremental Wet Deposition Over Background Within a Given Radius due to Operation of the BTGP (per cent)*

	Case	Radius (km)		
I:	NO_3^-	10	25	50
	Case 1: high	0	5.4	4.8
	Case 2: moderate	0	0.1	0.6
	Case 3: least	0	0	0.1
II:	H^+			
	Case 1: high	0	9.1	8.0
	Case 2: moderate	0	0.1	1.0
	Case 3: least	0	0	0.1
III:	pH			
	Case 1: high	0	0.04	0.03
	Case 2: moderate	0	0	0
	Case 3: least	0	0	0

* For III, units are pH.

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The following points summarize Table 13:

1. There is essentially no deposition of HNO_3 within 10 km of the plant, even under worst case assumptions.

2. The BTGP incremental effect on local nitrate deposition (relative to background) ranges up to a maximum of about 5.4 % on a long term average, and the effect is essentially the same out to 25 or 50 km. In view of the conservative approach adopted in these calculations, however, this figure is actually not expected to exceed 3 %.

3. The impact on H^+ deposition, as a percentage of background, could range up to 9.1 % on a long-term average, corresponding to a pH depression of 0.04 units. Again this is a conservative estimate and it is more likely that the annual average impact would be around 5 or 6 %. The corresponding pH depression of precipitation would be 0.02 units.

Incremental effects of BTGP on wet deposition at a certain distance from the plant are also included in Figure 11. Again, as was discussed in the previous section, the incremental effect at a certain distance is greater than the integrated effect within that distance, as given in Table 13, because there is no deposition within the induction period.

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The greatest potential effect on wet deposition of nitrate is expected to occur in the high deposition case at a distance of approximately 20 km. For typical operation of BTGP, as given in Section 3.1.2, it amounts to a 7.2 % increase averaged over long periods. The maximum incremental effect on hydrogen ion (relative to background) is 12 % as a long term average corresponding to a pH depression of 0.05 units. The figures quoted are all appropriate to the high deposition case, and should therefore be regarded as upper limits.

The foregoing results are derived from a rather simple model based on a number of assumptions. As mentioned earlier, the high deposition case results are extremely conservative and should be interpreted as such. The reasons for the conservativeness are given in detail below.

1. The individual parameters in Case 1, highest deposition, were all chosen to be simultaneously at the extreme of their probable range. (See Table 10 and the earlier discussion on model assumptions.)

2. The plume was assumed to be in contact with the ground at all times. This is usually not the case, and there can be no dry deposition from an elevated plume.

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3. As discussed in Section 3.3.4, most of the precipitation in the Vancouver area occurs in winter, with the most likely time of day for precipitation falling during the hours of darkness. However, the discussions of Sections 2.1 and 3.2 clearly indicate the dependence of the oxidation rate of NO on solar intensity, with the rates being reduced by, typically, a factor of 10 to 50 at night, in parallel with the reduction in OH concentration (Calvert, 1981). Thus, for much of the precipitation scavenging, a very much reduced value of the rate constant, k, would be appropriate.

4. No consideration has been given to the possible neutralization in the atmosphere of acidic species derived from BTGP emissions. That is, each molecule of emitted NO or NO_2 is assumed to result in the formation of one molecule of nitric acid, and thus one hydrogen ion. In reality, a significant fraction of the resulting nitric acid will be neutralized.

Confirmation that the predictions are conservative is obtained by the comparison given earlier between the predictions of this model and the experimental results obtained in St. Louis.

In summary, a number of approximate calculations were made to assess the potential impact of the BTGP emissions on the NO_3^- and H^+ deposition fields within 25 km of the plant. By considering that the

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emissions are wet deposited over a 90° sector to the west of the plant, the calculations suggest that the 0 to 1.4 % of BTGP's NO emissions scavenged over a year could increase the local NO_3^- deposition levels by, at most, roughly 5 % over background. It is felt that these numbers are quite conservative and the actual value is probably lower. In fact, if the moderate deposition case values are used (which appears more resonable for the long term conditions) the minor fraction of BTGP's annual NO emissions scavenged would cause an incremental deposition of less than 1 % within 25 km. In either case, i.e., the high deposition or moderate deposition case, BTGP's impact represents a small increase over a relatively small background deposition loading (compared to Peterborough, Ontario for instance).

Care should be taken not to interpret these model estimates too rigidly because they have not been verified experimentally. However, the small amount of experimental data available from Centralia and St. Louis, although not directly representative of conditions in the Greater Vancouver area, do provide some confidence in the model estmates.

Similar calculations for estimating the increased H^+ deposition due to BTGP emissions gave a range from less than 1 % for the moderate case up to 9 % for the high deposition case. The precipitation pH depressions due to these values of increased H^+ deposition ranged from 0 to 0.04 pH units (over the long-term).

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In summary, both the model predictions and the experimental evidence indicate that under clean background conditions, the BTGP emissions should have almost negligible effect on the local NO_3^- and H^+ deposition fields. Under extremely dirty background conditions caused by significant urban emissions, the BTGP emissions could add moderate amounts (i.e. up to 5 % for NO_3^- and up to 9 % for H^+) to the background urban deposition fields. However, these latter estimates are expected to be high and the actual incremental impact of BTGP emissions over current deposition levels is expected to be smaller. Also, at typical wind speeds, precipitation scavenging of BTGP emissions will probably not begin until they have travelled over and beyond the city and harbour areas of Vancouver. Hence, only Capilano Lake would be in the area most affected by BTGP emissions, although the effect is expected to be minimal.

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5. CONCLUSIONS

The results of a qualitative and semi-quantitative assessment of the potential impact of the operation of the BTGP on the local deposition of acidic compounds clearly demonstrate that the fraction of emissions wet and dry deposited locally would be small and would represent a very small incremental deposition above that taking place as a result of other GVRD emissions. The limited precipitation data which are available for Vancouver indicate pollutant deposition rates which are significantly lower than for other selected locations in Canada. Available data indicate that existing acid deposition rates have not caused changes in alkalinity and pH for a number of lakes in the GVRD. The incremental effect of BTGP emissions is therefore not expected to be significant.

Highly conservative assumptions indicate that the maximum incremental nitrate loadings in precipitation within 50 km of the plant could be up to about 5 % above the urban background and the corresponding incremental H^+ ion loadings up to about 9 % (corresponding to a depression of precipitation pH of 0.04 units). The major conclusions arising from the assessment are as follows:

- Because the primary emissions from BTGP, i.e. NO and NO_2, are not efficiently scavenged by wet or dry mechanisms, direct deposition of these species is not of concern.

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- The main atmospheric oxidation products of these emitted contaminants are nitric acid and peroxyacetyl nitrate. Both of these species are capable of causing the acidification of precipitation, and are relatively efficiently scavenged. However, essentially no formation of these secondary pollutants occurs within the first 40 to 60 minutes after emission. The corresponding distance at which production, and therefore deposition, commences depends on wind speed, and at average wind speeds, will take place over the city or the harbour of Vancouver during typical precipitation events.

  Conservative estimates of the additional contribution of the BTGP to long-term acidic deposition within 25 km of the plant are from 0 % to 0.4 % for dry deposition and from less than 1 % to 5.4 % for wet deposition. The effect is essentially the same within 50 km. If the BTGP contribution to acidity of precipitation is calculated, the predicted effect is from less than 1 % up to 9.1 % on a long-term basis. The corresponding pH depression is 0.04 units.

  These are very conservative estimates result from a careful choice of assumptions in the model calculation. Even so, the figures suggest that the incremental impact due to BTGP emissions is small. It is probable that the real impact would be less than these conservative calculations predict.

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Furthermore, the small increment due to BTGP emissions is above a background which is relatively small compared to other areas such as Southern Ontario.

- The fraction of BTGP emissions deposited as acidifying species within 50 km of the plant is small - 4 % at most. Less than 17 % will be deposited within 100 km.

- The areas of greatest impact of BTGP emission are expected to be in a 90° sector west of BTGP for wet deposition and two 90° sectors - one east and one west of BTGP, for dry deposition. Only Capilano Lake is in the area most affected by BTGP emissions but the effect is expected to be small.

- The deposition of pollutants derived from the BTGP emissions involves a complicated interplay between chemical conversion and wet and dry deposition. These, in turn, depend on many factors, which have been discussed in the body of the report, and which are as yet poorly quantified for the Vancouver area. It is clear that a more comprehensive data set on air and precipitation quality, which is currently not available, would be required to obtain a more precise quantitative assessment of the effects of operating the BTGP.

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APPENDIX 1

PRECIPITATION - ONLY WIND DIRECTION

(FROM EMSLIE, 1982).

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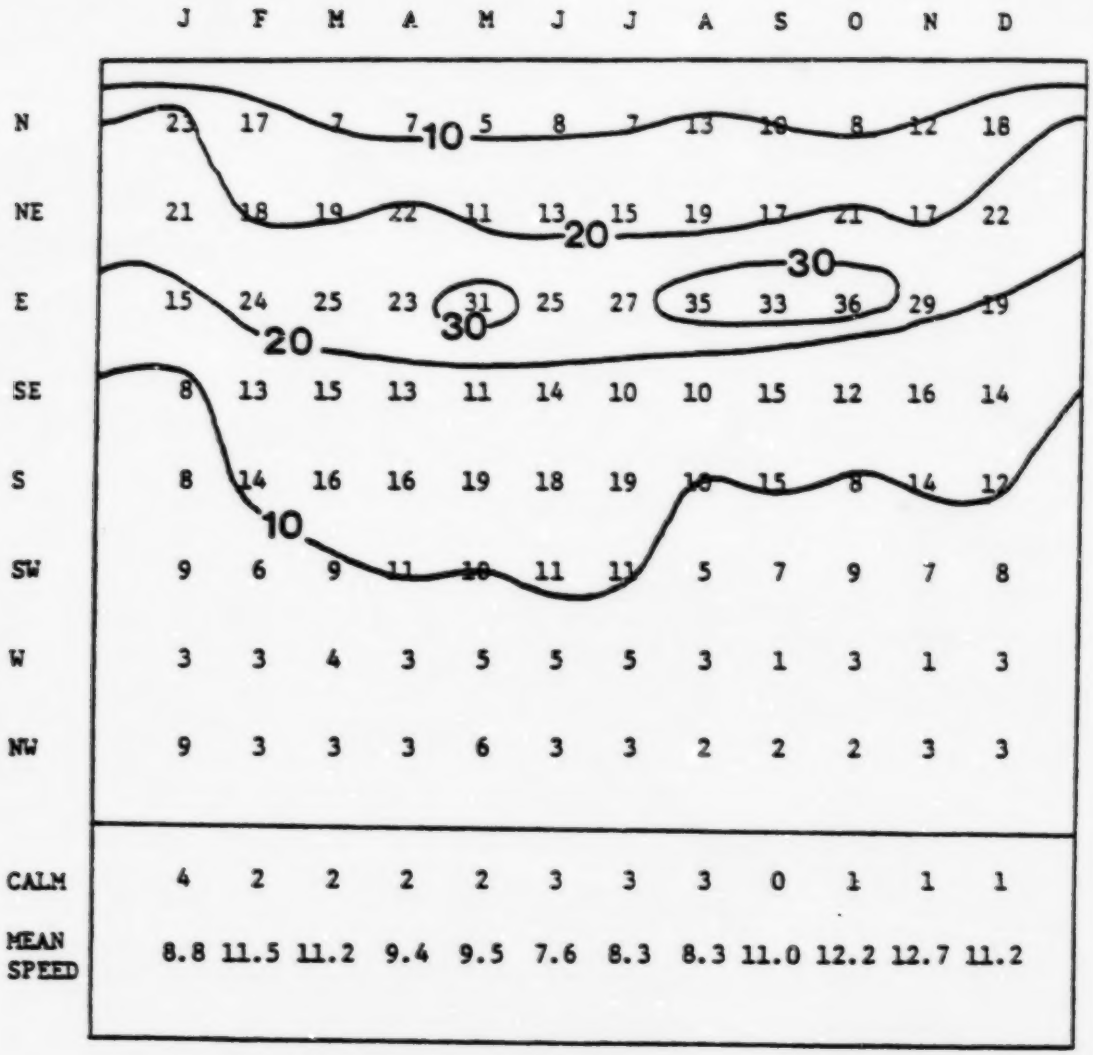

Burnaby Mountain BCH, B.C. - Percentage Frequency Wind Direction (and Calms) and Mean Wind Speed (MPH) by Month During Precipitation Hours (May 1973 - May 1978).

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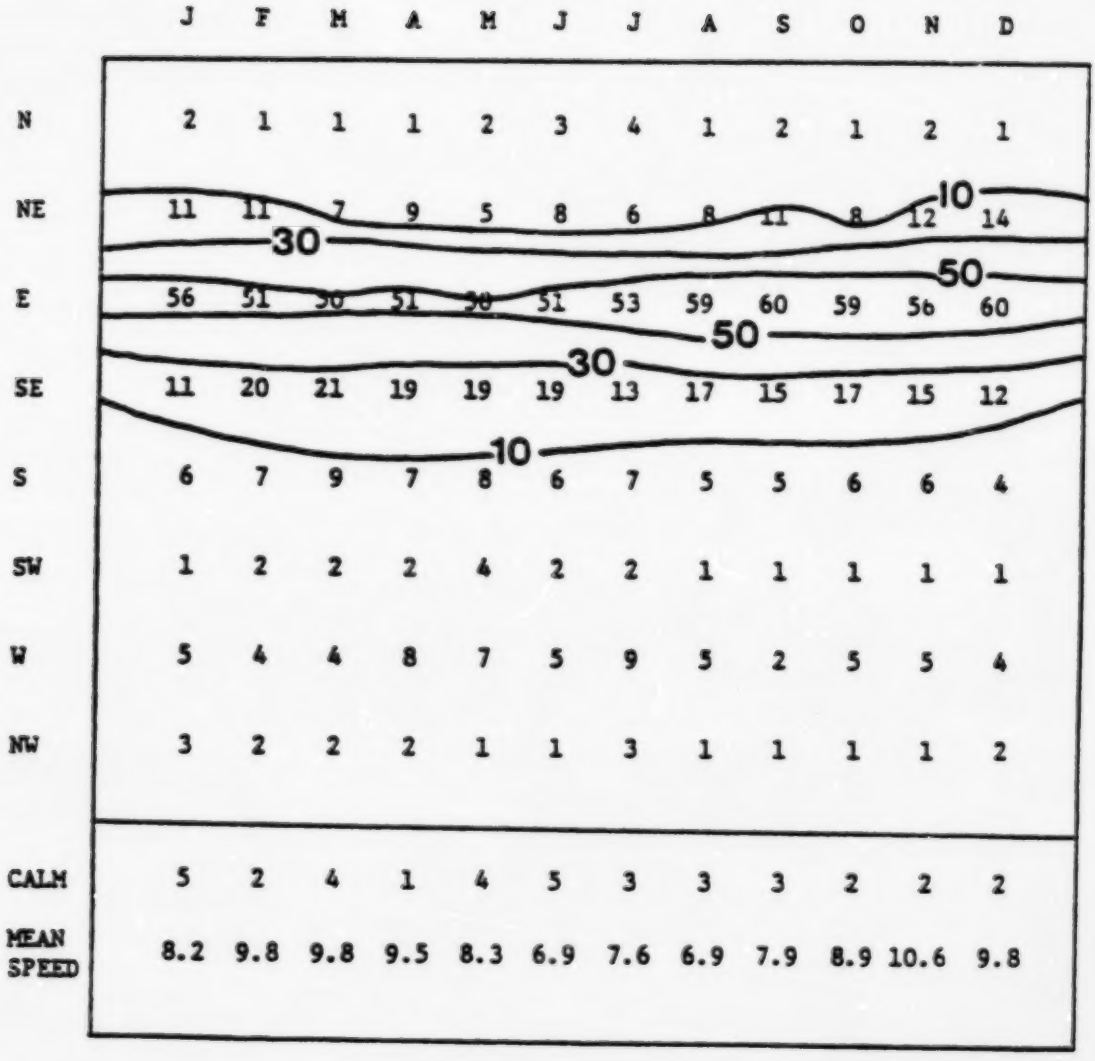

Vancouver International Airport, B.C. - Percentage Frequency Wind Direction (and Calms) and Mean Wind Speed (MPH) by Months During Precipitation Hours (May 1973 - May 1978).

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APPENDIX 2

SIMPLE BOX MODEL

This model is a variation of the model developed by Venkatram (1980).

Consider a point source to be emitting into a mixed layer of height z_i. The wind speed, u, in the layer is assumed to be constant and the horizontal spread of the plume is represented by a constant angle θ. The dry deposition velocity is given by v_d, NO to HNO_3 conversion is approximated by a first-order reaction occurring in one step with rate constant k and the scavenging coefficient for wet deposition is given by λ_w.

The assumption concerning NO conversion to HNO_3 is a drastic simplification and requires some justification. The rate constant, k, to be used in the model, was obtained by forcing the data of Table 1 to fit the first order rate law

$$[HNO_3] = [NO]_o \ (1 - e^{-kt})$$

where: $[HNO_3]$ = concentration of HNO_3 at time t

$[NO]_o$ = initial NO concentration

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In the case where conversion of NO to NO_2 is much more rapid than the rate of subsequent oxidation of NO_2, the latter step becomes rate determining, and the rate of the overall process will be accurately first order with a rate constant, k, equal to that of the second (slow) step. This is approximately the case for the "high" data of Table 3, and correspondingly the correlation coefficient for the force fit was 0.9985. The fit to the "low" data of Table 3 was nearly as good, with a correlation coefficient of 0.9946, thus justifying the simplification of the NO chemistry for the sake of this model.

The assumption of reaction in a single step implies negligible deposition of the intermediate NO_2. As was discussed in Section 2.4, both wet and dry deposition of NO_2 is known to be slow, particularly in comparison with the conversion reactions.

Given the assumptions above, then using a mass balance approach, the physical system can then be represented by the following differential equation:

$$\frac{d}{dr}(C_1 r) = -\frac{C_1 r}{u}\left[k + \lambda_{w1} + \frac{v_{d1}}{z_1}\right] \qquad (1)$$

where C = concentration (moles m^{-3})

r = distance from the source (m)

u = average wind speed (m s^{-1})

k = effective first-order rate constant for the NO to NO_3 conversion process (s^{-1})

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λ_w = scavenging coefficient (s^{-1})

v_d = dry deposition velocity ($m\ s^{-1}$)

z_i = mixed layer height (m)

and subscript 1 refers to NO.

Integrating (1) and assigning

$$C_1 = \frac{Q_1}{r\theta z_1 u} \quad \text{at } r/u = 0$$

where Q_1 = emission rate of NO in moles s^{-1}

and θ = horizontal spread of the plume (assumed as a constant angle)

gives

$$C_1 = \frac{Q_1}{r\theta z_1 u} \exp\left[- (k + \lambda_{w1} + \frac{v_{d1}}{z_i})(\frac{r - r_o}{u}\right] \qquad (3)$$

where $C_1\ (r_o) = \frac{Q_1}{r_o \theta z_i u}$

Here, r_o represents the distance from the source where (3) holds. It is, therefore, the distance at which the concentration can be represented by a well mixed box. For the purposes of this model, it is assumed that r_o is very close the stack. Under precipitating or

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inversion conditions this will not be totally accurate but it will be seen that at distances of 10 km or more, it will not affect the results of the model drastically.

If we now turn our attention from NO to the conversion product HNO_3, the following equation can be written

$$\frac{d}{dr}(C_2 r) = -\frac{C_2 r}{u}\left[\lambda_{w2} + \frac{v_{d2}}{z_i}\right] + \frac{kC_1 r}{u} \qquad (4)$$

Note that the subscript 2 refers to HNO_3 and concentrations are given in molar units. Then

$$\frac{d}{dr}(C_2 r) + \frac{C_2 r}{u}\left[\lambda_{w2} + \frac{v_{d2}}{z_i}\right] = \frac{k}{u}\frac{Q_1}{\theta z_i u}\exp\left[-(k + \lambda_{w1} + \frac{v_{d1}}{z_i})\frac{r}{u}\right] \qquad (5)$$

If we assume that the wet scavenging rate and the dry deposition velocity for NO are negligible (reasonable assumptions), then we can let

$$\beta_1 = \frac{1}{u}(k + \lambda_{w1} + \frac{v_{d1}}{z_i}) = \frac{k}{u}$$

$$\beta_2 = \frac{1}{u}(\lambda_{w2} + \frac{v_{d2}}{z_i})$$

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and $\phi = C_2 r \; ; \; A = \frac{kQ_1}{u^2 \theta z_i}$

Then (5) can be rewritten and integrated to radius r from the stack in the form

$$\frac{d}{dr}\phi + \beta_2 \phi = A \exp(-\beta_1 r) \tag{6}$$

to give

$$\phi = \frac{A \exp(-\beta_2 r)}{(\beta_2 - \beta_1)} \exp(\beta_2 - \beta_1) r - 1] \tag{7}$$

where r_o again is set approximately equal to 0. Therefore, the concentration HNO_3 at any distance from the stack can be written as

$$C_2 = \frac{kQ_1 \exp(-k\, r/u)}{u\theta z_i r(\lambda_{w2} + \frac{v_{d2}}{z_i} - k)} [\exp(\lambda_{w2} + \frac{v_{d2}}{z_i} - k)r/u - 1] \tag{8}$$

With equation (8) we can now assess the impact of the BTGP on wet and dry deposition in the area of the plant. If we consider F to be the fraction of time that wet or dry deposition can occur, i.e.

$F = F_w$ for wet deposition

$= F_d$ for dry deposition,

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then the total wet and dry deposition, D_w and D_d respectively, to a given radius r, can be determined by integrating the following equations:

$$D_w = \int_0^r F_w \lambda_{w2} z_i \theta C_2 r dr \qquad \text{(moles } s^{-1}\text{)} \qquad (9)$$

$$D_d = \int_0^r F_d v_{d2} \theta C_2 r dr \qquad \text{(moles } s^{-1}\text{)} \qquad (10)$$

The resulting total deposition normalized by emission rate is

$$\frac{D_{w2}}{Q_1} = \frac{F_w \lambda_{w2} k}{u^2 (\beta_2 - \beta_1)} \left[\frac{1}{\beta_1} [1 - \exp(-\beta_1 r)] - \frac{1}{\beta_2} [1 - \exp(-\beta_2 r)] \right] \qquad (11)$$

and

$$\frac{D_{d2}}{Q_1} = \frac{F_d v_{d2} k}{u^2 z_i (\beta_2 - \beta_1)} \left[\frac{1}{\beta_1} [1 - \exp(-\beta_1 r)] - \frac{1}{\beta_2} [1 - \exp(-\beta_2 r)] \right] \qquad (12)$$

Equations (11) and (12) provide the basis for the model calculations presented in Section 4.

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